

APPENDIX B: EXPANDED DISCUSSION OF SELECTED TOPICS

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B-1. THE SALTWATER INTRUSION: EFFECTS ON STRATIFICATION AND STABILITY

HISTORICAL PERSPECTIVE

Data on salinity from Station 522 as a function of depth, spanning a period of nearly 40 years, are given in Fig. B-1.1. These are the results of three studies, including the present one. The data from the earlier publications (Smith and Thompson, 1927; Collias and Seckel, 1954) are not as complete as those from the Metro study, but the period of maximum annual stratification (August-November) is delineated in each case.

It can be seen that maximum annual bottom salinities today are only 7 to 10 percent as high as for the period 1925-1953. Also, oxygen profiles drawn concurrently provide evidence that hypolimnetic oxygen depletion is less severe today than previously. The curve for September 5, 1974 corresponds to the lowest oxygen values measured between 10 and 15 m for the year. All other curves in the figure represent oxygen levels at the time of the maximum bottom salinities, and for these conditions indicate appreciably higher hypolimnetic oxygen concentrations during 1974 than during the previous studies. If this is indeed true, then it seems likely that it was due largely to stratification supported by the characteristically high bottom salinities. The lowest annual salinity values recorded for the depth interval 14-15 m were 3.2 and 5.8‰ for the 1927 and 1954 studies, respectively. It appears that complete flushing did not occur at any time during these years. On the other hand, the corresponding value for the present investigation at Station 522 was .028‰ with a maximum of .060‰ salinity for the first six months of the year. Seckel's data (Seckel, 1953) indicate that, except for one 2-month period, the lake hypolimnion was anoxic from January, 1951 through September, 1953 (the entire period of the study).

This considerable reduction in the volume of saline water flowing into the lake was due to more frequent operation of the saltwater siphon that pumps inflowing saltwater to the seaward side of the locks and to the installation of a saltwater barrier at the Salmon Bay end of the large lock in 1966. Continuous siphoning alone had reduced salinities in Lake Union to 1974 levels by 1960 (data from Driggers, 1964). Previously, the locks had dumped a daily net average in summer (in excess of the saltwater siphon capacity) of $1.23 \times 10^5 \text{ m}^3$ of diluted seawater into the Salmon Bay catch basin (Seckel, 1953). With the siphon in operation, the basin filled in four to five days; with the siphon closed, only one day was required. All water in excess of this

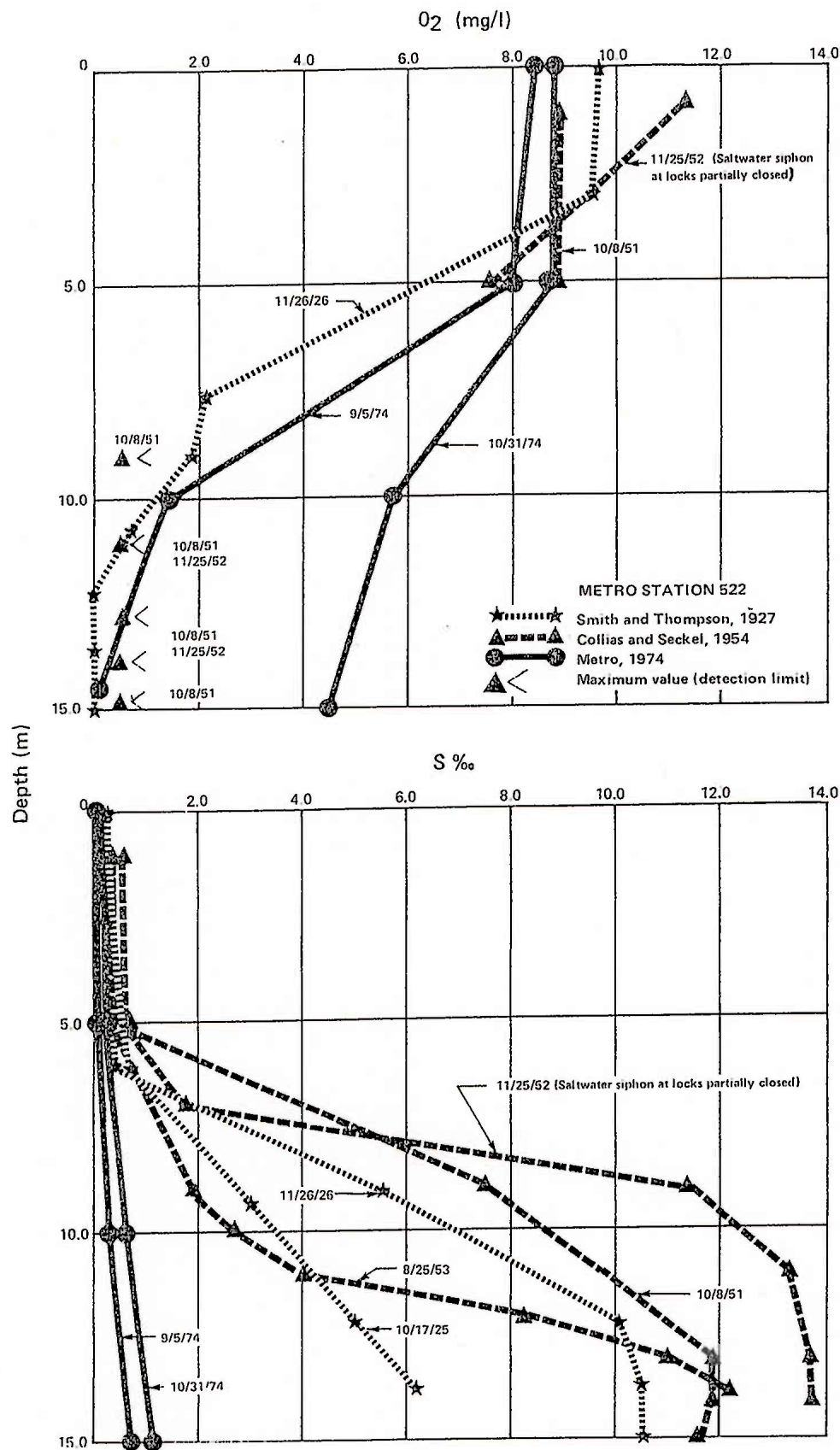


Fig. B-1.1. Historical depth profiles of salinity and dissolved oxygen at Lake Union Station 522.

amount flowed along the bottom of the Fremont Channel, over the shallow sill, and into Lake Union. At a constant rate and with no flushing, calculations indicate this input would fill Lake Union to the depth of the interface (average 11 m) in about 380 days.

REDUCTION OF THE THREAT TO THE LAKE WASHINGTON OVERTURN

If we consider the effects of flushing and annual periods of drainage sufficient to prevent saltwater intrusion through the Fremont Channel, the net effect in past years was that of a large saltwater catch basin annually filling to the brim and threatening to spill into Lake Washington. An excess of saltwater in the hypolimnion of Lake Washington would intensify the stratification, possibly preventing overturn in the fall. The result would be the stagnation of the brackish water accumulating at the bottom, with disastrous consequences for the ecology of the lake.

In the early autumn of 1952, a major saltwater intrusion into Lake Washington was detected (Seckel, 1953). Calculations showed that it would subsequently require the lake 4 to 5 years to reduce its salt content to the 1950 level, provided no additional salt entered during the interim. Seckel also estimated that the 1952 intrusion brought into the lake five times the amount that could be introduced without causing a net annual increase. A salt budget comparison with a stagnant meromictic European lake indicated that Lake Washington could not afford to have so large a salt input any more often than every 4 to 5 years.

On the basis of the data in 1974, it is apparent that the danger of major saltwater intrusions into Lake Washington was reduced considerably by the previously mentioned modifications of the Chittenden Locks. The 1952 intrusion occurred when the halocline rose to the 7 m depth on November 14. A comparison of annual maximum salinity values at this depth taken from stations in the northeastern sector of Lake Union (Stations 52-10 and Metro 532) (Fig. D-1) show a ratio of 15.8:1 for 1952:1974. On the basis of these figures and Seckel's calculations, Lake Washington could tolerate $15.8 \div 5.0 = 3.2$ times as much saline water from the 1974 Lake Union hypolimnion as it received from that of 1952 with no net annual salt increase. The threat of net annual salt increases in Lake Washington have been reduced since 1952 by approximately one minus the ratio of the 1974 and 1952 sill depth salinities. On the basis of maximum salinities in 1974 from depths >7 m, this reduction factor ranges from 67 to 94 percent. The respective minimum rates of discharge from the Chittenden locks during the fall months were approximately

2.3 and 3.4 m³/sec for 1952 and 1974. Both values are considerably below the minimum discharge (50 m³/sec) said by Seckel to prevent salt intrusion into Lake Union.

PHYSICAL DYNAMICS OF THE PRESENT SYSTEM

By mid-June of 1974 the rate of runoff discharge through the Fremont Canal had fallen below the critical level of 50 m³/sec specified by Seckel (1953) as the minimum outflow required to prevent saltwater intrusion into Lake Union (see Fig. B-1.2). Subsequently, a slight bottom salinity increase was detected at all stations on July 18 (Fig. 4-4), and a major intrusion occurred at Stations 518, 522 and 532 between August 15 and September 5. From this information, it is evident that the saltwater catch basin in Salmon Bay required a minimum of one month to fill, the actual value depending on its initial saltwater level in mid-June. Since the time of flow between the basin and the lake is less than 5 h, on the basis of Seckel's data, the rate of saltwater flow into the basin in 1974 was a maximum of 17 percent of that in 1953. Seckel's estimate of a maximum basin fill time of 5 days in 1953 was assumed in this calculation. For the 2-month interval between mid-June and the first major intrusion into the lake, the same analysis yields a 1974 to 1953 ratio of 8 percent. This value is close to the ratio of maximum salinities at the bottom of the lake for these two years.

In the foregoing calculations a very abrupt and short-lived increase in salinity that occurred at the bottom at Station 518 in early July was not taken into consideration. This value was verified by an independent conductivity aliquot taken from the same field-sampling apparatus. Examination of daily use records from the Hiram Chittenden Locks indicated no significant lockage increase during the preceding 4-month period (Fig. B-1.2). If we assume its existence, then the anomalous saltwater bolus was apparently quite small and inconsequential to lake dynamics.

Reference to Figs. 4-4 and 4-5 shows the influence on stratification of a short period of relatively high winds between September 5 and 19. All four stations were profoundly influenced to a depth of 10 m. Initially, the influence was evident in a concurrent decrease in salinity and increase in oxygen at Stations 518 (bottom, at 11 m) and 522 (10 m) and a slight increase in oxygen at the bottom (14.5 m) at Station 522. The effect at Stations 526 and 532 was not immediately apparent but was delayed for up to 2 weeks. Sufficient data on water circulation are not presently available for a plausible explanation for this observation.

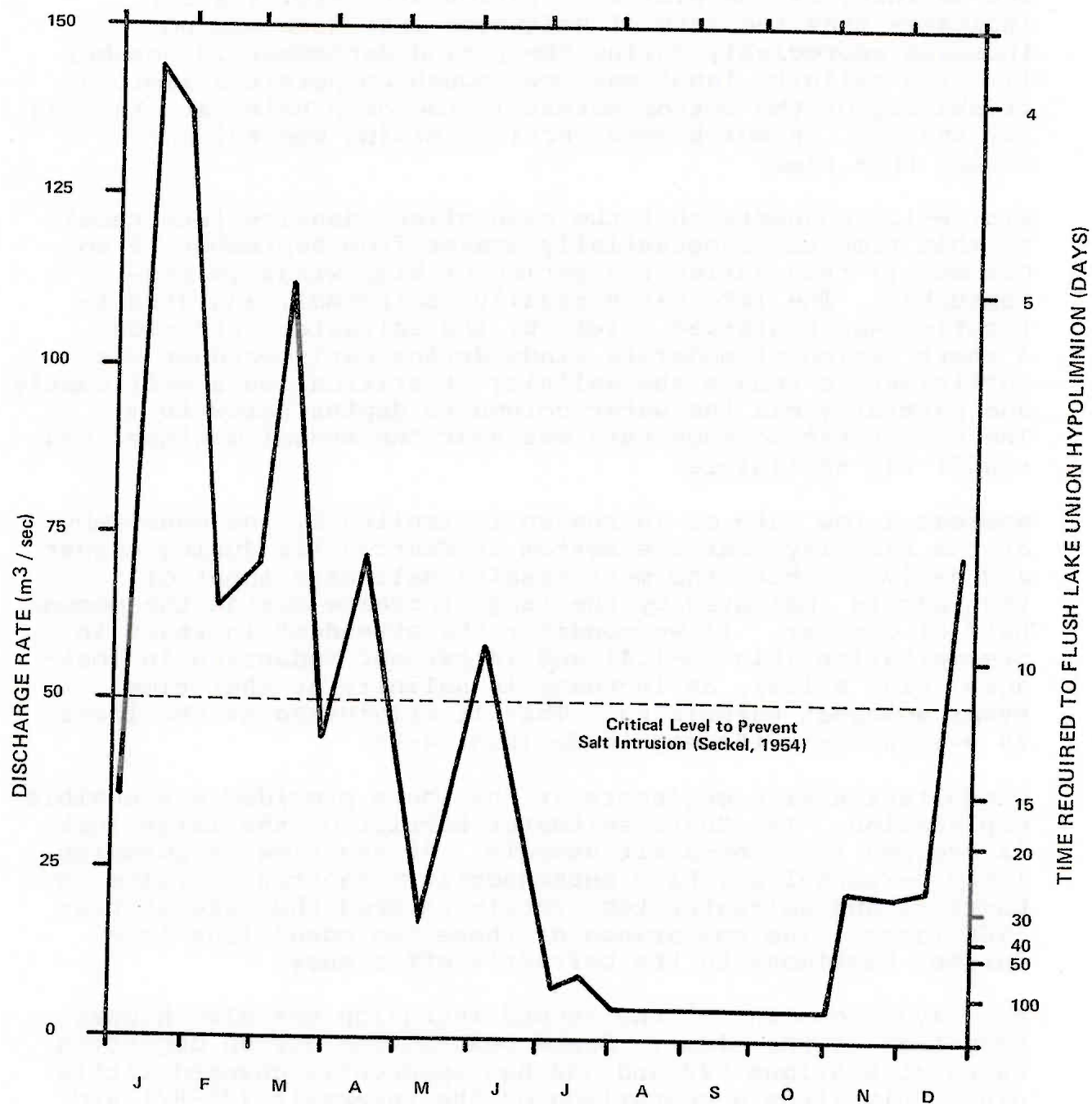


Fig. B-1.2. Rates of water discharge from the Hiram M. Chittenden (U. S. Government) Locks, 1974. U. S. Army Corps of Engineers data.

The salinity-oxygen plot for Station 518 (Fig. 4-4{a}) indicates that the rate of saltwater intrusion did not increase appreciably during the period September 19-October 17. The salinity input was low enough to permit a limited freshening of the bottom waters in the deep holes at Stations 522 and 532, in which some vertical mixing was evident during this time.

Fig. B-1.3 suggests that the pycnocline (density interface) at that time was substantially weaker from September 19 to October 17 than during the period of high winds in mid-September. The lake was virtually isothermal, and stratification was sustained solely by the saltwater intrusion. A short period of moderate winds during early October was sufficient to reduce the salinity stratification significantly and partially mix the water column to depths below 10 m. The concurrent lockage rate was near the annual maximum, and runoff was negligible.

Whereas a low rate of intrusion is implied by the constancy of the salinity near the bottom at Station 518 during August and early October, the most massive saltwater input of the year is indicated by the large increase during the second half of October. If we consider the attendant increase in precipitation (Fig. B-1.4) and 14 percent reduction in lockages (Fig. B-1.2), an increase in salinity at that time seems somewhat surprising. That it originated at the locks is evident from its magnitude (Fig. 4-2).

Consultation with engineers at the locks provided a plausible explanation. The 20-ft saltwater barrier on the large lock is dropped for deep-draft vessels. At the time in question, locks personnel may have subsequently forgotten to raise the barrier, and saltwater temporarily entered the lake at pre-1966 rates. The comparison of these two conditions is a further testimony to the barrier's efficiency.

The oxygen content of the second intrusion was also higher than that of the first. Since near-bottom oxygen depletion rates at Stations 522 and 532 had apparently changed little since July (from a comparison of the intervals 7/5-8/1 and 10/31-11/14), the considerably higher oxygen concentrations resulting from the second intrusion may reflect a significantly higher saltwater flow rate and/or volume (i.e., a shorter and/or less effective period of oxygen depletion prior to entry into Lake Union).

In the same vein, it is interesting to note that Fig. 4-2 shows that salinities at the bottom at Station 518 were lower than at Stations 522 and 532 for the first intrusion

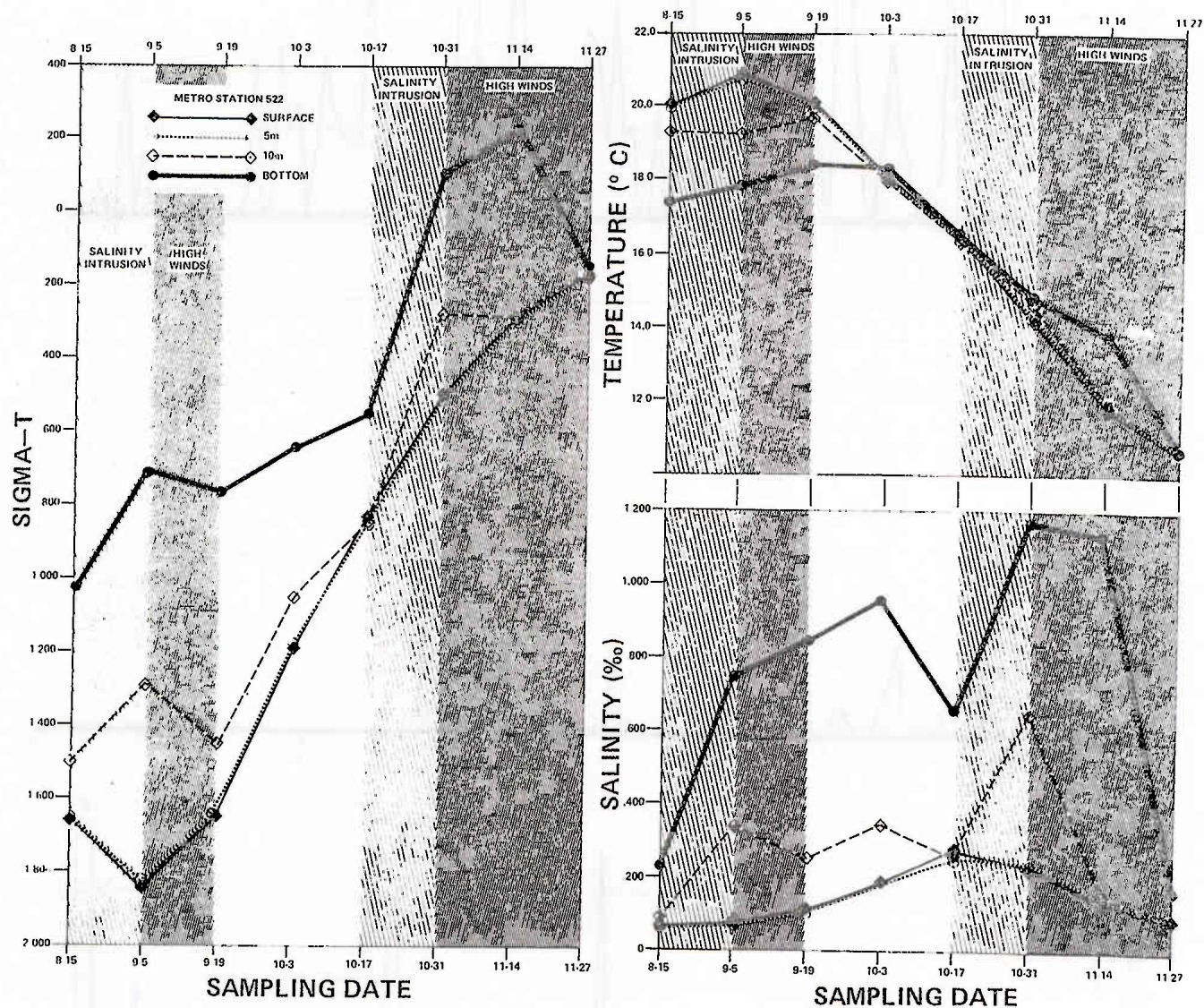


Fig. B-1.3. Salinities, temperatures and sigma-t's at the surface, 5 m, 10 m, and bottom of Metro Station 522 in Lake Union, 8/15 to 11/27, 1974.

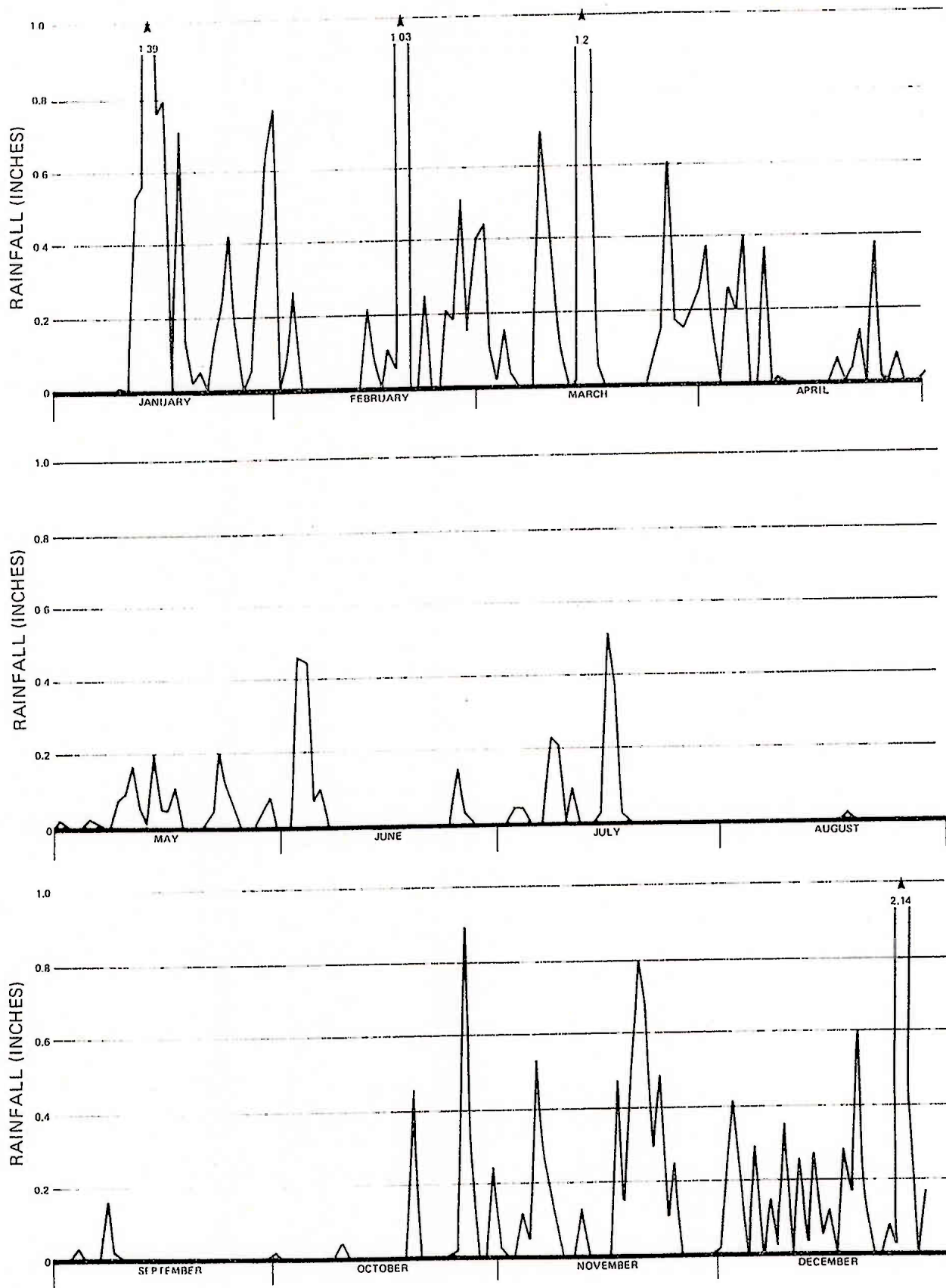


Fig.B-1.4. Rainfall in the Seattle region, 1974. Data from the U.S. National Weather Service, Seattle-Tacoma International Airport.

but were similar to those at these stations for the second. The near-bottom saltwater jet described by Seckel (1953) may well have been thin enough during the first intrusion for the maximum core salinities to escape detection by the sampling methods used. However, if the second entry had a thicker jet, that would have increased the probability of sampling of the core water. These deductions are in keeping with observations of vertical circulation made by Driggers (1964), who found through dye-dispersion studies that a stratified layer as thin as 20 cm can exist in the hypolimnion of Lake Union for several days.

The saltwater intrusion in October prolonged the stratification of the lake by nearly one month. As is manifest in Fig. B-1.3, the water column was mixed to deeper than 10 m by October 17 and required perhaps two additional weeks of similar conditions to mix to the bottom of the deepest areas. On October 19 for 15 h, winds were in excess of 7.5 knots; this force is indicated by the trends to have been potentially enough to complete the destratification process. The introduction of saltwater, however, raised the pycnocline to above 10 m. Beyond that time, four weeks of strong winds, together with increased runoff and reduced lockage, were required to mix and flush the bottom waters at Stations 522 and 532.

Although the lake overturn was virtually complete by late November, a pool of relatively unmixed water flowed into the basin at Station 522 during early December (see Figs. 4-1, 4-2, and 4-3). Two weeks later, no traces of it remained. This water was high in phosphate, ammonia, iron and salinity and very low in dissolved oxygen. In fact, with the exception of a somewhat lower temperature, it had properties characteristic of bottom water taken in mid-September from Stations 522 or 532. Its existence is confirmed by independent measurements of conductivity and oxygen by the U. S. Army Corps of Engineers (Seattle District, 1974).

The phenomenon was similar to that observed under similar circumstances by Seckel (1953) at Station 52-10 (Fig. D-1). In that instance, the intruding bolus was assumed to be residual bottom water draining from elsewhere in the lake. However, whereas Station 52-10 lay in a trench (see Fig. 2-1) capable of supplying water from similar depths, Station 522 was an isolated hole. Parcels of dense bottom water would have had to come from the eastern side of the lake and flow intact over a 4- to 5- ft. barrier. Obviously, there is much to be learned about the circulation of Lake Union.

On the subject of circulation, one final comment seems appropriate. The preceding discussion indicates that the movement of near-bottom water masses between Station 518 (the

"gateway" to the lake) and Station 522, for example, required a maximum of 2 weeks. This flow rate corresponds to a minimum of about 6 m/h, which approximates the 8.9 m/h southerly rate estimated by Driggers (1964) for the lake hypolimnion.

B-2. THE SALTWATER INTRUSION: EFFECTS ON THE PO_4/Fe CYCLE IN THE HYPOLIMNION

The chemistry of the PO_4/Fe sediment trap is further complicated by the presence of the saltwater intrusion. Estuarine studies indicate that increasing salinity favors the desorption of the phosphorus from particulates, or breakdown of the iron-inorganic phosphorus complex (Upchurch et al., 1974). Further, Carritt and Goodgal (1954) found the particulate uptake of phosphorus from seawater with salinities of 17 and 34 percent to be less than in freshwater but the reduction not in direct proportion to the salinity.

However, there is evidence that comparatively low salinities do not significantly affect phosphate adsorption and desorption. Studies on sediments from Lake Mendota, Wisconsin (Williams et al., 1970), provide appropriate data on the particulate adsorption and desorption of inorganic phosphorus in a 0.1M NaCl medium (roughly equivalent to a salinity of 6.4%). For phosphate levels analogous to those in 1974 in Lake Union bottom waters, extrapolated adsorption and retention values were 100 percent and 90 percent respectively. Lake Mendota is eutrophic and 22 m deep and has sediment total phosphorus and organic carbon levels comparable to those of Lake Union (refer to discussion on sediments, Section 5). Although the Lake Mendota cultures were unnaturally agitated, so that the adsorption and desorption processes were magnified, the relative difference (the retention) should be reservedly applicable to natural environments.

It thus seems likely that the present saltwater intrusion into Lake Union has little direct effect on phosphate release from the sediments. However, at the higher salinities entering the lake prior to the locks modifications, this effect may have been more significant. Part of the phosphate thus released may have subsequently been flushed out through the locks in the fall.

At this point, it seems pertinent to examine the potential contribution of the saltwater intrusion to the phosphate concentrations observed in the lake hypolimnion. From average near-surface salinity values recorded for central Puget Sound during August, 1974 (Metro data), an approximate dilution factor of 38:1 was calculated for saline bottom water reaching Lake Union Station 522 on September 5. Accordingly, a representative mean orthophosphate concentration for the source water (E. E. Collias: 1975 Metro Puget Sound Interim Studies data) would have been diluted to .002 mg/l, or about .5 percent of the observed value at Station 522.

This estimate pertains only to the contribution of orthophosphate from the seawater itself. It is not to say that the addition of nutrients to the saltwater once inside the locks was necessarily negligible. According to previous calculations, the saline water may remain in the locks catch basin for 1 to 2 months. This is ample time for the anoxic solubilization and accumulation of additional phosphate. Accordingly, these and any other nutrient additions during the estimated 5-h ship canal transit time could be carried into Lake Union.

Before in-lake modification by mixing or desorption processes, the initial orthophosphate concentrations in the intruding saltwater should represent the total orthophosphate accumulated during its transit through the ship canal. These values are listed in Table B-2.1: they exemplify possible intrusion maxima. The 1974 in-lake sampling was too infrequent for us to guarantee a total absence of prior in-situ increases. The data for August for Station 518 were omitted as nonrepresentative of the intrusion (see discussion, previous section).

Table B-2.1 Minimum Orthophosphate
Concentrations Observed
in the 1974 Saltwater
Intrusions Into Lake Union

Stn.	Depth (m)	Data	Orthophosphate (mgP/l)
518	11.0	10/31	.024
522	14.5	8/15	.076
		10/31	.043
532	15.0	7/18	.079
		10/31	.066

From the data, it can only be said that the question of a high orthophosphate contribution from the saltwater intrusion is further intensified. The input should be investigated as part of any study of the nutrient budget for Lake Union. Whereas this phosphate is not likely to be immediately available for algal uptake, its ultimate fate is uncertain because of the complexity of phosphate cycling within the lake.

Storm and sanitary waste overflows are another potentially major source of algal nutrients in Lake Union. Fortunately, it is the nature of the hydrological system that the majority of such overflows occur when the lake flushing rate is high and other conditions are not conducive to primary productivity. Their impact on algal production is therefore probably diminished considerably from what it would be

otherwise. However, combined sewer simulation models by the City of Seattle Engineering Department indicate that overflows frequently occur during the spring bloom in the lake (see loading estimates in Section B-4). It is therefore highly likely that wastewater nutrients generally contribute to the total production. Analyses of treatment plant influent and studies of urban storm drainage currently being done by Metro should soon make possible reasonably accurate quantitative estimates of the waste nutrient contributions.

A satisfactory evaluation of the various aforesaid aspects of the Lake Union phosphate budget may prove easy compared with answering questions on vertical transport and phytoplankton availability. One such question, asked by many and reiterated by Mortimer (1971), concerns the extent to which the various forms of phosphate cycling in the hypolimnion are available for biological production in a lake. How much of the phosphate released under anoxic conditions is actually available for algal uptake at the density interface and remains available throughout the water column after the overturn? Also, to what extent would this source continue to support algal blooms after the cessation of phosphate input to the lake?

Lake Union and Lake Washington might prove to be significantly different in this respect, for in contrast to that of the larger lake, the lower boundary of the photic zone in Lake Union is below the pycnocline (density interface), where the phosphate transfer to the surface waters must take place. Wind mixing alone would thence disseminate any available phosphate throughout the zone of active productivity.

B-3. THE SALTWATER INTRUSION: EFFECTS ON THE CONCENTRATIONS OF DISSOLVED METALS.

Zinc concentrations near the bottom during the period of stratification tended to be high relative to those at other depths. This observation suggests the possibility of influence by the saltwater intrusion. A plot of the zinc concentrations vs salinity near the bottom for all stations appears in Fig. B-3.1. The solid line was fit to the data by curvilinear regression. The dashed lines are conservative estimates of the 68 percent confidence limits (± 1 standard deviation). The data points corresponding to $S_{\text{‰}} \leq .10$ (the shaded area in the figure) were excluded from this calculation. As evident from the considerable range of the associated zinc concentrations, these points include appreciable interference from particulate zinc mixed into the water column under conditions of low stability.

At least two inferences may be made from this information: (1) the zinc concentration of the intruding saltwater is low ($\leq .05$ ppm for $S_{\text{‰}} \leq 29$), and (2) the increased salinity instigates a release of zinc by the lake sediments. The first conclusion is apparent from the magnitude of the observed zinc concentrations and the nonlinear relationship between the two parameters. The second seems valid as an interpretation of the nonlinear increase of Zn concentrations, so that negligible values of $\Delta \text{Zn} / \Delta S_{\text{‰}}$ are attained at higher, source-level salinities.

Zinc is a highly adsorptive metal that is easily scavenged from solution by particulate matter at pH levels ≥ 7.0 (Tomlinson, 1970; Tomlinson and Renfro, 1972). Similarly, it is easily released by ionic exchange with charged, dissolved species that are present in higher concentrations and/or for which the surface adsorption sites have a greater affinity. In the present instance, the adsorbed zinc could have been replaced on the sediment surfaces by exchange with undetermined ions present in the saltwater in (relatively) high concentrations. The maximum zinc concentration of .05 ppm (at $S_{\text{‰}} = 29$), for seawater entering the system through the locks) represents an approximate maximum concentration presently obtainable in this system by ion exchange. Within Lake Union itself, this value should not exceed .025 ppm Zn unless a major malfunction of the lock siphon and/or saltwater barrier occurred.

Rohatgi and Chen (1975) demonstrated a similar release of zinc, copper, nickel, and lead from experimental mixtures of seawater and suspended particulates taken from the Los Angeles River. The equilibrium losses of these trace metals were found to be 60, 66, 72, and 16 percent, respectively and relatively independent of suspended solids concentrations. It seems likely that the sediment concentrations of these four metals are similarly affected by the saltwater intrusion into Lake Union.

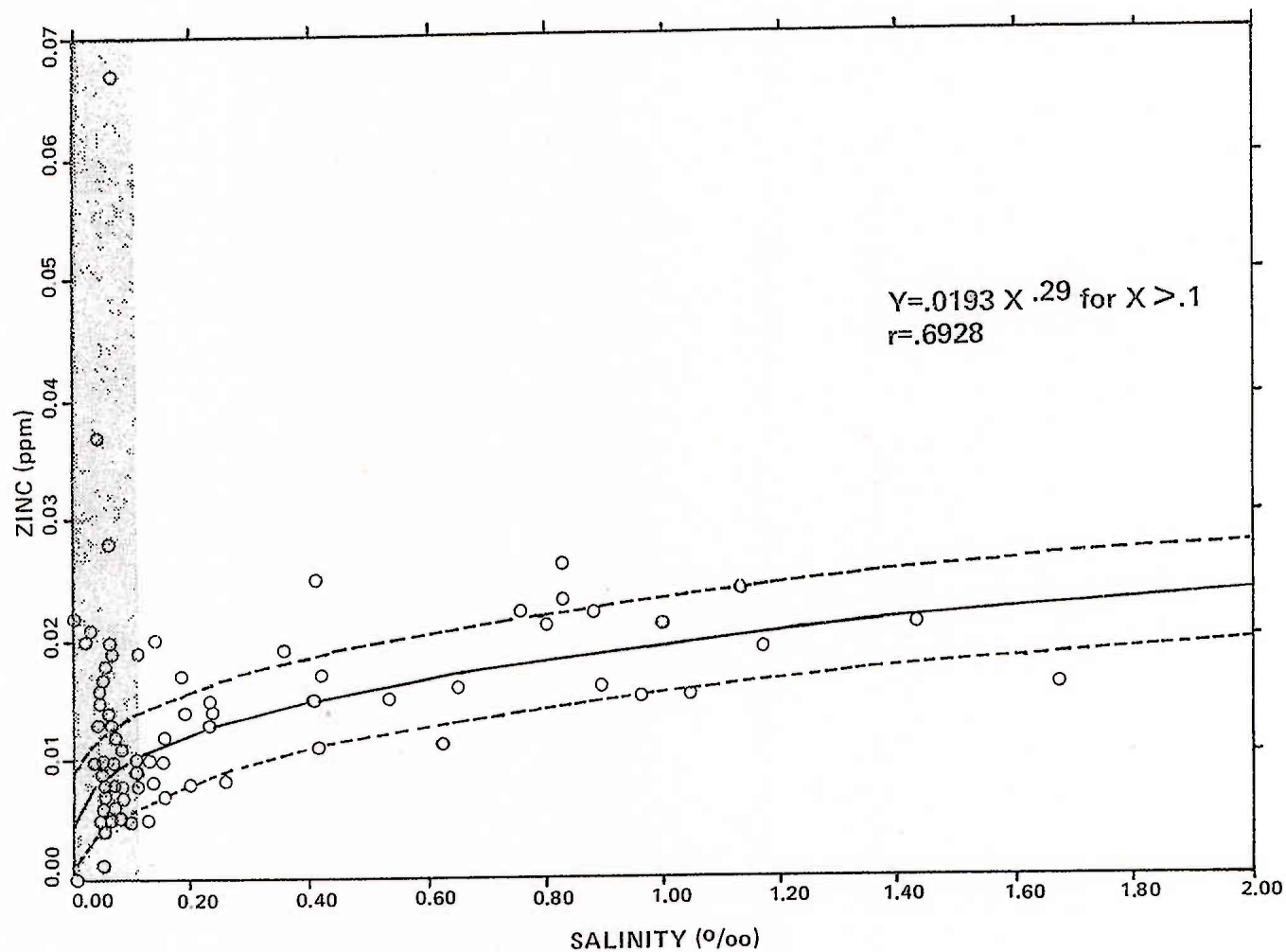


Fig. B-3.1. Zinc vs salinity for the bottom waters of Lake Union, 1974. Dashed lines represent standard error of the estimate.

The real importance of this apparent desorption of metals from the sediments is not manifest in the present concentrations of the dissolved fraction, even though the zinc values are within the range of maximum concentration recommended by EPA for fish protection (Table 4-2). Rather, it lies in the potential for even higher concentrations of similar origin that might occur with further contamination of the sediments. Adsorption and desorption of metallic ions such as zinc, copper, nickel, and lead are functions of ion concentration. Under otherwise similar conditions, higher surface concentrations promote higher dissolved concentrations upon desorption. Thus, particles that have concentrated such metals through adsorptive processes may release biologically detrimental concentrations when flushed by a more saline medium. The ultimate level of water toxicity will depend in part on the degree of sediment contamination. As detailed in Figure 5-1, the latter condition is worsening in Lake Union.

B-4. THE NITROGEN AND PHOSPHORUS BUDGETS: QUANTITATIVE ESTIMATES OF SELECTED COMPONENTS.

GENERAL

The significance of the cyclic fluctuations in nitrate content can perhaps better be explained with reference to Fig. B-4.1, which summarizes the 1974 nitrate + nitrite profile data for a representative in-lake station (522). The profiles for December 26 and January 3 through March 1 demonstrate the buildup of nitrate + nitrite levels in the lake during the period of high runoff and complete wind mixing. Several sources contribute to this increase: terrestrial runoff, dustfall (including rainfall), sewer overflows, and nitrification in the water column and sediments. Compared with the spring maxima in Lake Washington in 1974 (Pieterse, 1974), the concentrations of inorganic nitrogen compounds in Lake Union are very similar. This fact would point to runoff (inflow from Lake Washington) as the major contributing source.

Although there is a paucity of data on fluvial nutrient input for recent years, rough estimates of the nitrogen contributed by stream runoff and rainfall can be made on the basis of past data. In 1957, the total inorganic nitrogen loading by stream input to Lake Washington was 2.26×10^5 kg $\text{NO}_3 + \text{NO}_2 -\text{N}$ (Edmondson, 1970). If we consider the estimated volume of Lake Washington ($2.99 \times 10^9 \text{ m}^3$: Barnes and Schell, 1973) and rain ($8.0 \times 10^7 \text{ m}^3$: National Weather Service data) inputs, the stream loading of inorganic nitrogen in 1957 increased the "mixed-lake" concentration by $.05 \text{ mg NO}_3 + \text{NO}_2 -\text{N/l}$. If this input is accepted as representative of conditions in 1974, it would account for 13 percent of the observed spring maximum in $\text{NO}_3 + \text{NO}_2$ concentrations in Lake Union. Since the assumption of total mixing in Lake Washington is not entirely valid, the actual input of $\text{NO}_3 + \text{NO}_2$ from this source must be somewhat higher. There are no streams directly tributary to Lake Union.

An estimate of the input of nitrogen from dustfall can be made on the basis of data presented by Johnson et al. (1966). From 3-month totals for nitrate deposits from dustfall in the Lake Union area and average monthly concentrations for the entire region, the annual fallout on Lake Washington is calculated at $980 \text{ mg NO}_3 -\text{N/m}^2$ and on Lake Union at $830 \text{ mg NO}_3 -\text{N/m}^2$ each year. These inputs would correspond to an increase of $.021 \text{ mg NO}_3 -\text{N/l}$ in the "mixed-lake" concentration of Lake Washington and of $.002 \text{ mg NO}_3 -\text{N/l}$ in the "mixed-lake" concentration (including volume added by runoff and rainfall) of Lake Union.

Lake Union would also be indirectly affected by the fallout on Lake Washington, since part of it would subsequently be advected into the smaller lake through the Montlake Channel.

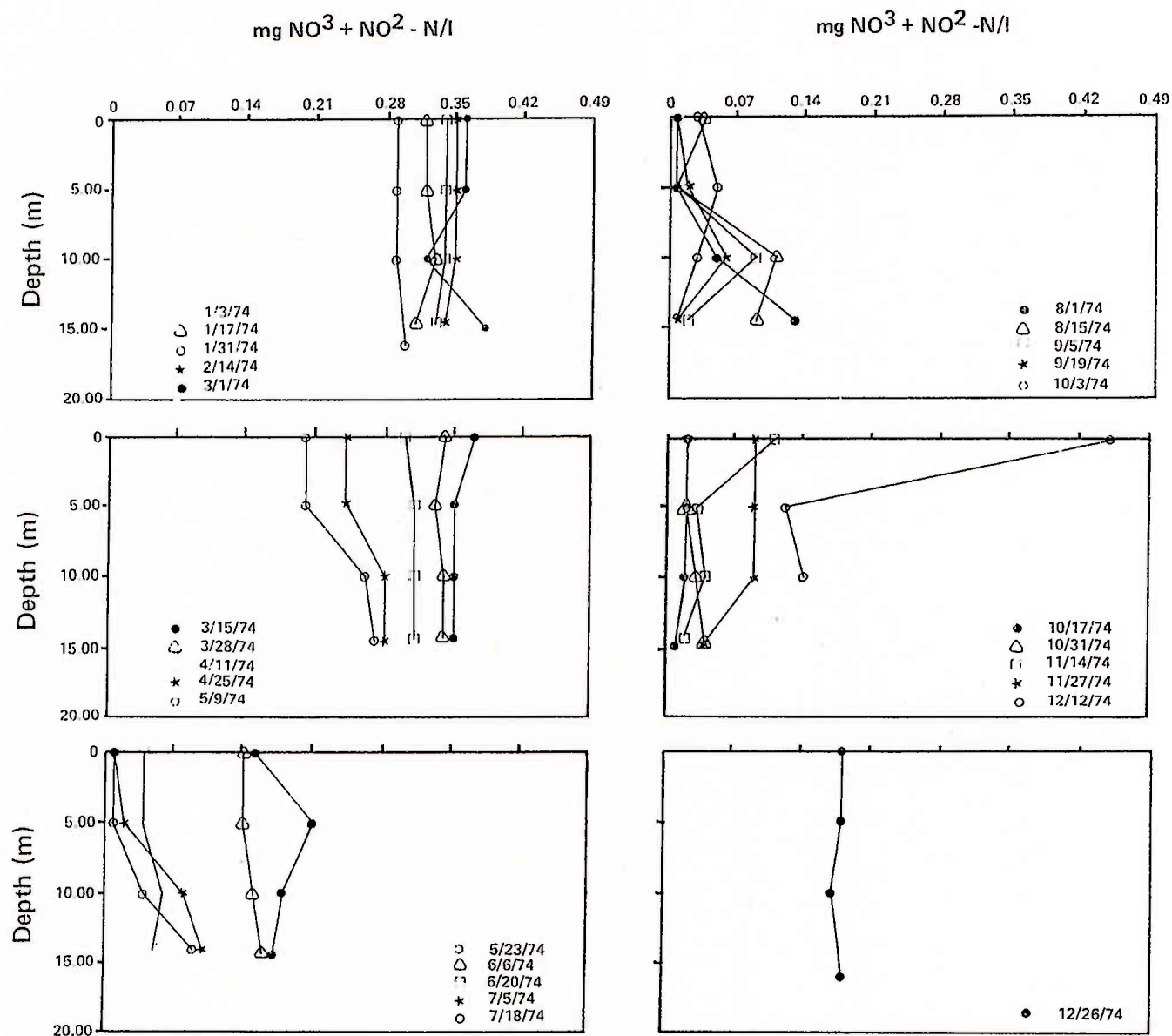


Fig. B-4.1. Depth profiles of nitrate + nitrite from Lake Union Station 522, 1974.

The winter-spring (mid-November to mid-March) fraction of the total fallout on Lake Union would amount to only about 1.5 percent of the 1974 spring maximum. The annual combined phosphate fallout for the two lakes may add a total of .005 mg PO₄-P/l, or 2.5 percent of the average annual surface concentration in Lake Union, most of which would be input to the surface of Lake Washington.

From these estimates, it is apparent that a large fraction of the winter and spring nitrogen loading to Lake Union is contributed by other sources. The largest portion is undoubtedly released by the bottom sediments. Fillos and Swanson (1975) recorded an average release rate of 5 mg NH₃-N/hr/m² in laboratory experiments with lake sediments. Further, Austin and Lee (1973), using stirred, batch cultures, confirmed the rapid conversion of ammonia and the much slower conversion of soluble Kjeldahl nitrogen to nitrate in Lake Mendota sediments under aerobic conditions (refer to the discussion on phosphate for a summary of Lake Mendota's relevant characteristics). Solubilization of Kjeldahl nitrogen likewise proceeded under anaerobic conditions, but was accompanied by release of ammonia. They also found that inorganic nitrogen was released seven times as fast from sediments under aerobic conditions as under anaerobic conditions. The conversion by sediment bacteria of ammonia and soluble organic matter to nitrates and nitrites effectively mobilizes large quantities of sediment nitrogen and releases it into the water column. Unlike orthophosphate released by sediments under anaerobic conditions, the nitrogen liberated under aerobic conditions is known to be available for biological production.

NUTRIENT LOADING BY WASTEWATER OVERFLOWS

Sewer overflows are a potentially major source of both nitrogen and phosphorus loading to Lake Union. Until recently, reasonable estimates of contributions to the lake from overflows were not possible, owing to an absence of data on input volume. The City of Seattle Department of Engineering (George Hsieh) has now completed a series of such volume estimates, using computer modelling techniques. In the most comprehensive version, rainstorms occurring over a period of 11 years were incorporated, in which a minimum rainfall rate of .03 in/hr was observed for at least 5 min. As Fig. B-4.2 shows, this model included outfalls 125, 126, 175, 128, 130, 132, 133 and 135. For purposes of the present calculations, we extrapolated the mean volume data from these outfalls to cover the remainder of the combined non-Metro outfalls in the lake: 123, 144, 146, and 147.

The remaining outfalls are storm drains, emergency overflows, or individually monitored components of the Metro system, as the pipe at Galer Street. The inputs from storm drains have

been considered separately here. Five of these lines drain up to 250 acres of the I-5 freeway next to the lake. Their volume contributions may be estimated from simple data on area and rainfall, and an assumed runoff coefficient of .90 (suggested by K. Harris, City of Seattle Engineering Dept.). Complete data on volume of overflow for Metro's Galer Street outfall are available through direct computer monitoring. Emergency overflows operate only during electrical or mechanical failure and therefore have a negligible impact.

Mean concentrations of wastewater nitrogen and phosphorus species entering natural waters in the Seattle region are summarized in Table B-4.1. The sources of these data are listed in the table footnotes. From the most representative figures from this table and the wastewater volume estimates outlined above, the mean annual nitrogen and phosphorus loadings from wastewater to Lake Union were calculated; these are summarized in Table B-4.2.

From these figures it may be seen that the forms of nitrogen and phosphorus that are usable by phytoplankton (orthophosphates and polyphosphates; nitrate, nitrite, and ammonia) are added to the lake by overflows in nearly equal concentrations. Storm water contributes almost as much usable nitrogen as combined wastewater does, but only about 9 percent as much usable phosphorus, approximated here by the total hydrolyzable phosphate fraction. Most of the wastewater nitrate + nitrite enters the lake through the storm drains.

Localized concentrations of these nutrient contaminants in Lake Union are difficult to assess at present, owing to a lack of data on circulation. At best, we can examine the relative effects of the flushing by water running through the Ship Canal in 1974, as opposed to a hypothetical no-flushing situation. Assuming complete mixing and a constant rate of overflow, we have calculated the final nitrogen (year end) and phosphorus concentrations; the results are given in the last two columns of Table B-4.2. From these data, it becomes strikingly apparent that the massive annual flushing of Lake Union is extremely important to the maintenance of its water quality. Any significant reduction of flow through the Lake Washington Ship Canal could permit serious buildups of algal nutrients in Lake Union. As will be demonstrated later, the same holds true for heavy metals contamination.

It must be strongly emphasized here that the previous analysis represents a hypothetical calculation that is useful in a relative sense only. The actual environmental concentrations of nitrogen and phosphorus from wastewater depend on circulation patterns around the individual outfalls. In this sense, they may range from the values listed in Table B-4.2 for total mixing with 1974 flushing rates to the estimated

Table B-4.1. Mean Concentrations of Selected Nitrogen and Phosphorus Constituents of Combined and Storm Wastewater Entering Natural Waters in the Seattle Region

Parameter (mg/l)	COMBINED			STORM	
	CATAD/Dexter ⁽¹⁾	Lake Wash. ⁽²⁾	URBD/CBD ⁽³⁾	I-90 ⁽⁴⁾	I-5 ⁽⁵⁾
TPO ₄ -P	1.35 ⁽⁶⁾			.38 ⁽⁶⁾	
THPO ₄ -P		2.15 ⁽⁶⁾	2.26		
OPO ₄ -P			.70 ⁽⁶⁾		.26 ⁽⁶⁾
Total N			13.0	1.99	
Total Org. N			8.65	.64	
NH ₃ -N	.75 ⁽⁶⁾	1.49	3.46	.26 ⁽⁶⁾	.17
NO ₃ +NO ₂ -N	.12 ⁽⁶⁾	1.13	.94	1.09 ⁽⁶⁾	1.37

(1) Metro data, unpublished. Dexter Regulator, Lake Union.
Sampling dates: 12/18/74, 12/20/74, 1/3/75, 1/5/75, 1/12/75 3/1/75.

(2) Dalseg and Leiser, 1970. Station 4903, Henderson St. overflow.
Sampling dates: 10/9/69, 10/24/69, 10/27/69, 10/29/69,
11/28/69, 12/8/69, 12/10/69, 12/19/69-12/22/69, 1/9/70,
1/13/70, 1/16/70-1/20/70.

(3) Metro data, unpublished - support data for RIBCO, 1974.
Seattle Central Business Dist. Sampling dates: 3/10/73,
3/16/73, 6/6/73, 8/16/73, 9/19/73.

(4) Metro data, unpublished - support data for Farris et al., 1973.
E. end of Mercer Is. Floating Bridge and S. Bellevue Inter-
change. Sampling dates: 11/18/72, 11/23/72, 11/24/72,
1/23/73, 1/29/73.

(5) Dalseg and Farris, 1970. Section of I-5 freeway between
College and Holgate Streets. Sampling dates: 2/17/70,
3/2/70, 3/3/70, 3/6/70.

(6) Values used for further analysis in the present study.

Table B-4. 2. Estimated Annual Loadings of Nitrogen and Phosphorus
From Combined and Storm Wastewater Entering Lake Union Directly through Outfalls

Parameter	City Outfall (1) Discharge (10 ³ Kg)	Metro Outfall (2) Discharge (10 ³ Kg)	Storm Drain (3) Discharge (10 ³ Kg)	Total Discharge (10 ³ Kg)	Final Concentrations (8) Assuming Total Mixing	
					1974 Flushing (4) (mg/l)	No Flushing (5) (mg/l)
Total PO ₄ -P (6)	2.05	.071	.313	2.43	.002	.099
Total Hydrol. PO ₄ -P	3.27	.113				
Ortho PO ₄ -P	1.06	.037	.215	1.31	.001	.053
Total Inorg. N (7)	1.32	.045	1.12	2.48	.002	.101
NH ₃ -N	1.14	.039	.215	1.39	.001	.057
NO ₃ +NO ₂ -N	0.18	.006	.900	1.09	.001	.044

- (1) Volume data calculated from City of Seattle Engineering Dept. computer model (see text for details). Mean no. of overflows/outfall/yr=32.4. Mean volume/overflow=3.90x10³ m³. Total annual volume contributed by 12 outfalls = 1.52x10⁶ m³.
- (2) Volume data for Galer St. (Dexter Regulator) outfall - 1974, 1975 average. Mean no. of overflows/yr=11.0. Mean volume/overflow=4.77x10³ m³. Total annual volume contributed by outfall = 5.25x10⁴ m³.
- (3) Volume calculations based on assumed runoff coeff.=.9 for 36 inches rain/year; runoff from estimated 250 acres of pavement (I-5) (K. Harris, City of Seattle Engineering Dept.) Total annual volume contributed by 5 storm drains = 8.3x10⁵ m³.
- (4) Volume calculations based on assumed 1.30x10⁹ m³ annual flow through the lake.
- (5) Volume calculations based on assumed absence of outflow and absence of inflow other than combined and storm wastewater. Ave. volume of Lake Union = 2.45x10⁷ m³.
- (6) Values low owing to use of persulfate digestion (O'Connor and Syers, 1975).
- (7) Computed as sum of NH₃ and NO₃+NO₂.
- (8) See text for explanation.

outfall concentrations given in the previous table. Other important considerations include the suitability of environmental conditions for algal blooms at the time of overflow and the physical state (soluble or particulate) of the introduced nutrients. Even under high-flow conditions, particulates may settle to the bottom of the lake, further decompose, and become available for algal uptake through future sediment-water cycling. In this light, organically bound particulate nitrogen and phosphorus would also have to be considered for complete budget estimates.

It must also be mentioned that the accuracy of the figures in Table B-4.2 is largely dependent on the accuracy of the Hsieh overflow model. Preliminary field measurements show good agreement, but the lower threshold set for rainstorms may still be a little too high, as indicated by close agreement between the number of rainstorms actually used and the resultant number of overflows estimated. If so, then the nutrient (and heavy metals) loading estimates made in this report are somewhat conservative.

There is also some overlap in the estimates, however. Since outfalls 126 and 175 are nominally freeway storm drains that receive some sanitary wastes downstream, they were included in the analyses for both the combined and the storm wastes. The error resulting from any consequent duplication is believed to be small.

Presumably minor sources of nitrogen and phosphorus include ground-water seepage and nonsewered runoff directly to both lakes. These contributions require indirect estimates that are outside the scope of the present analysis.

NUTRIENT REGULATION IN THE WATER COLUMN

Relationships between orthophosphate-phosphorus and inorganic nitrogen in the water column during the major phytoplankton blooms are given in Figure B-4.3. The data have been plotted sequentially so that the station-to-station trends will be clearly delineated. During the active growth phase (4/11-5/23), nitrogen and phosphorus were utilized in an approximate ratio of 15:1. Immediately prior to this time, phosphorus was available in excess. The 15:1 ratio was established following uptake and storage of this "luxury phosphorus" as polyphosphate (Kuhl, 1974), a form not measured by the present method. Stored polyphosphates can be utilized by phytoplankters when orthophosphate concentrations in the water column become limiting.

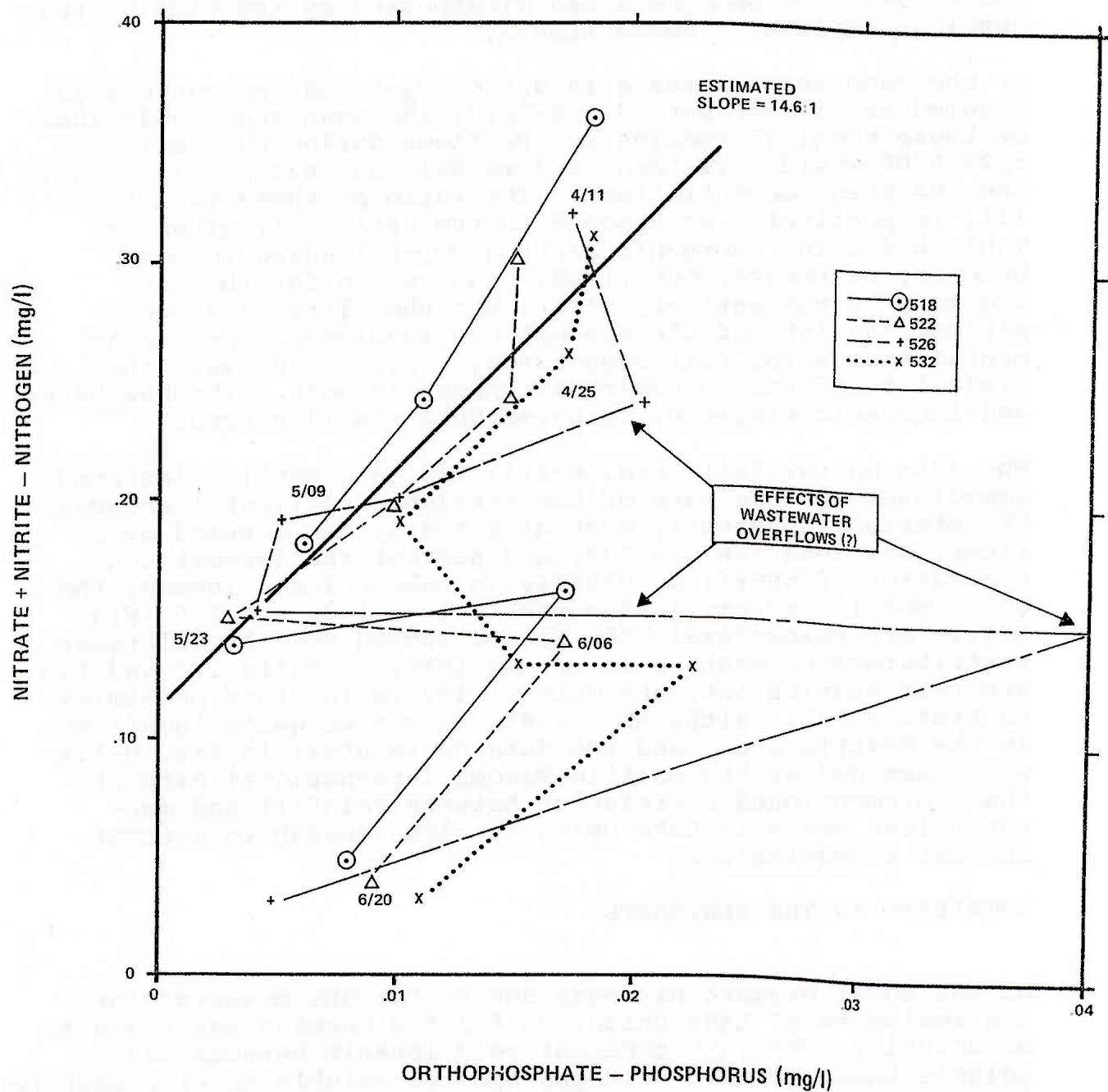


Fig. B-4.3. Nitrate + nitrite vs orthophosphate in the Lake Union water column, 4/11 to 6/20, 1974. Position of heavy line estimated.

The apparent effects of wastewater overflows on nutrient availability are also shown in Figure B-4.3. On April 25 and again on June 6, there were substantial increases in dissolved orthophosphate concentrations. In each instance, the excess orthophosphate had disappeared by the time of the ensuing sampling, 2 weeks thence.

If the same mean decrease in ortho $\text{PO}_4\text{-P}$ and inorganic N is assumed as for the period 5/9-5/23, the mean gross additions of these chemical species by overflows during the period 5/23-6/06 would have been .023 mg N/l and .024 mg P/l to give the observed concentrations. The ratio of these values is 1:1, respectively, as opposed to the ratio 2:1, given in Table B-4.2 for inorganic N:ortho $\text{PO}_4\text{-P}$ loading to Lake Union by wastewater overflows. The reason for the discrepancy is not entirely clear, but the difference seems minimal in light of the assumptions involved. The fundamental reason for this comparison, i.e., to increase the visibility of the concomitant changes in both orthophosphate and inorganic nitrogen, is nevertheless well served.

The data on rainfall (Fig. B-1.4) indicate small rainstorms immediately before each of the mentioned nutrient increases. The stations apparently most affected by the assumed overflows, Stations 526 and 532, are nearest the largest concentration of overflow outfalls in Lake Union. Indeed, the Hsieh model predicts that outfalls 128, 132, and 175 (Fig. B-4.2) are respectively the first, second and third largest contributors of wastewater to the lake; outfalls 128 and 175 are near Station 526, and outfall 132 is in close proximity to Station 532. Although rainstorms can be quite localized in the Seattle area, and the data on rainfall in Fig. B-1.4 were recorded at the Seattle-Tacoma International Airport, the aforementioned correlation between rainfall and macronutrient peaks in Lake Union is close enough to warrant further investigation.

NUTRIENTS IN THE SEDIMENTS

Of the total organic nitrogen concentrations measured for the sediments of Lake Union, 35.6 ± 6.0 percent was found to be soluble. The very coherent relationship between the soluble total organic nitrogen and the soluble total phosphate in the lake sediments (see Fig. B-4.4) seems to indicate that naturally occurring organic matter is the prevalent source of both. Indeed, Keeney et al. (1970) found that over 98 percent of the nitrogen measured in the sediments of each of 17 Wisconsin lakes was organic. A predominance of inorganic phosphate would be expected to give a much more

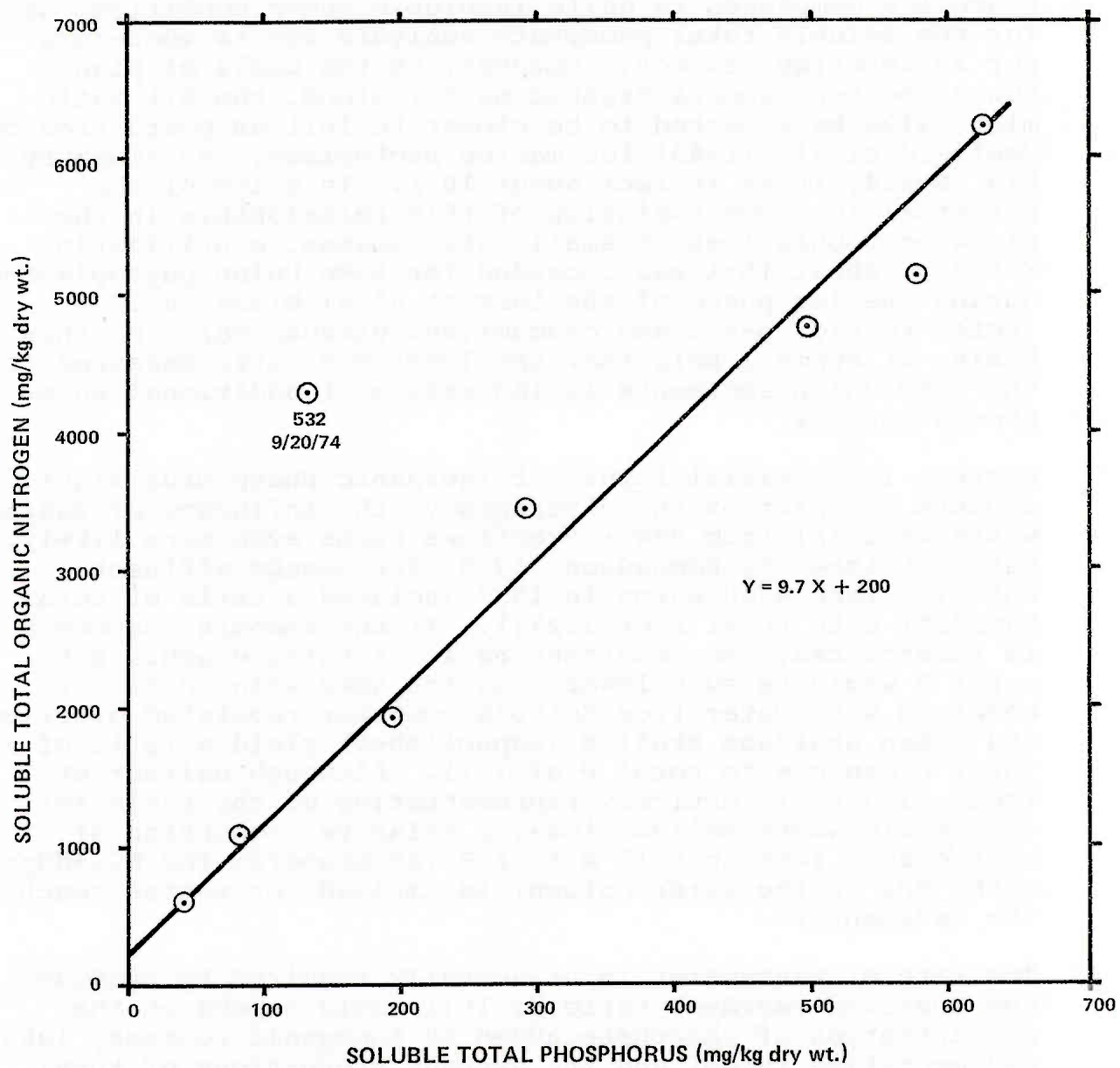


Fig. B-4.4. Soluble total organic nitrogen vs soluble total phosphorus in the Lake Union sediments, 1974. Position of line estimated.

variable relationship than actually observed, particularly since both stratified and unstratified water conditions are represented by these data. Phosphate found in ferric hydroxide complexes is quite insoluble under conditions used for the soluble total phosphate analysis and is therefore not an important factor. However, on the basis of planktonic and terrigenous organic matter alone, the N:P ratio might also be expected to be closer to 16:1 as postulated by Redfield et al. (1963) for marine protoplasm. As shown by Fig. B-4.4, it is in fact about 10:1. In spite of the potential for some variation of this relationship in the plankton populations of small water masses, a utilization ratio of about 15:1 was recorded for Lake Union phytoplankton during the log phase of the largest algal bloom in 1974 (refer to Fig. B-4.3 and concomitant discussion). On this basis, it seems likely that the lower N-P ratio measured for the Lake Union sediments is indicative of additional phosphorus sources.

Whereas interstitial inputs of inorganic phosphorus might account for part of the discrepancy, the influence of sanitary waste material from sewer overflows seems even more likely. Data published by Edmondson (1969) for sewage effluent entering Lake Washington in 1957 included a ratio of total kjeldahl N to total P of 3.27:1. If the ammonia component is subtracted, the resultant ratio of total organic N to total P would be even lower. In the same vein, data for combined wastewater from Metro's computer-regulated overflows and urban drainage studies (unpublished) yield a ratio of total organic N to total P of 12:1. Although neither of these figures is entirely representative of the ratio for settleable waste solids alone, a relative proportion of appreciably less than 15 N to 1 P (as measured for $\text{NO}_3 + \text{NO}_2 + \text{NH}_3$: ortho PO_4 in the water column) is implied for wastes reaching the sediments.

The rate of wastewater input actually required to produce the observed sediment ratio of 10:1 would depend on the concentration of phosphate added by inorganic sources, lake sedimentation rates, and the average proportions of total organic nitrogen and total phosphorus in the settleable wastewater solids. These factors are identified as important unknowns that must be evaluated in future studies of nutrient budgets for Lake Union. In addition, it is ultimately even more critical to assess in such studies the potential availability of the soluble sediment nutrients for primary production.

COMPARISON OF THE NUTRIENT BUDGET WITH THAT OF LAKE SAMMAMISH

Generally the primary goal in the study of nutrient budgets is the estimation of the availability of nutrients for

primary production. In this task, it is important to keep in mind that lake sediments provide a vast buffering capacity that helps to maintain the availability of nutrients from one growth cycle to the next. A decrease in periodic inputs of nitrogen and phosphorus from external sources upsets the concentration equilibria within the lake, and the sediment reserves are drawn upon for restoration of the balance. The subsequent rate of sediment nutrient depletion depends on effective washout rate, sedimentation, and lake morphometry (Vollenweider, 1969).

In this sense, lakes may differ considerably. For example, Emery (1972) noted that the recovery of Lake Sammamish from a state of high algal productivity after sewerage diversion was much slower than that of nearby Lake Washington had been. The slower recovery was attributed to the seasonal cycling of the phosphate between the hypolimnion (and sediments) and the surface waters. The phosphate is coprecipitated with colloidal ferric oxides within a few months following the fall overturn. Owing to increasingly anoxic conditions in the hypolimnion, the ferric iron is then reduced to the soluble ferrous state, and the adsorbed phosphate is liberated. Overturn of the water column later reverses this process, and the reformed ferric oxides are carried upward into the surface waters, together with the summer's accumulation of dissolved phosphorus. This phosphorus is again accumulated by the ferric oxides and remains available for algal uptake in the photic zone only as long as the associated particulate iron remains in suspension. The sustention of this state is a function of wind mixing and therefore of the lake morphometry.

A similar ferric oxide-phosphate "trap" exists in Lake Union, as previously noted. Coprecipitation with ferric hydroxide was probably responsible for the abrupt decrease in surface phosphate concentrations observed in late January at all stations (Fig. 4-10). It may also have caused the mid-October phosphate peak following a brief period of wind mixing of the increasingly unstable water column.

Though similar in some aspects of the phosphate budget to Lake Sammamish, Lake Union differs in some important aspects from it. It is considerably shallower, so that the wind mixing of phosphate, nitrate, ammonia, and particulate iron into the surface waters is more effectively promoted. Also, relative to its volume, Lake Union is flushed much more rapidly. The extremely high rate of flushing (annual average: one lake volume per week) is undoubtedly the single most important factor preventing a considerably higher level of contamination. This observation is tied closely to the

third major difference between the two lakes: Lake Union is much more urbanized and receives significantly higher concentrations of wastewater pollutants. On the whole, then, termination of wastewater discharges into Lake Union and the ship canal would probably result in a faster decrease of water and sediment nutrient concentrations than has occurred in Lake Sammamish and perhaps even Lake Washington.

B-5. THE HEAVY METALS BUDGETS: QUANTITATIVE ESTIMATES OF CONTRIBUTIONS FROM WASTEWATER

An analysis of heavy metals inputs to Lake Union from combined and storm wastewater was carried out in the same manner as for nitrogen and phosphorus (refer to the previous section). The selected mean data are presented in Table B-5.1 and the resultant loading estimates in Table B-5.2. The ranges offered for stormwater metals are artifacts of lower detection thresholds, which obscured single-figure estimates for some of the supporting measurements.

As it was for nitrogen and phosphorus, the importance of flushing by runoff is very strongly apparent from a comparison of the last two columns of Table B-5.2. An absence of flushing would result in one year in the accumulation of metals concentrations equaling or exceeding the EPA guidelines listed in Table 5-4. Lead is the metal discharged in the highest concentrations, and in nearly equal amounts by combined and storm wastewater. The loading of copper, zinc, chromium, and cadmium is higher from combined wastewater. Again, it is not feasible to estimate the magnitude of localized buildups of these contaminants without more extensive circulation studies.

The concentrations of the samples analyzed for Table B-5.1 represent the totals of the soluble and particulate fractions. The proportion contributed by each fraction is presently unknown, but the relatively high sediment concentrations indicate a large particulate fraction and/or subsequent scavenging by settling particulates in the lake itself.

**Table B-5.1. Mean Concentrations of Selected Heavy Metals
in Combined and Storm Wastewater
Entering Natural Waters in the Seattle Region**

Parameter (mg/l)	COMBINED		STORM
	CATAD/Dexter ⁽¹⁾	URBD/CBD ⁽³⁾	I-90 ⁽⁴⁾
Cu	.14 ⁽⁵⁾	.49	.07 - .08 ⁽⁵⁾
Pb	.89 ⁽⁵⁾	.35	1.59 - 1.67 ⁽⁵⁾
Zn	.51 ⁽⁵⁾	.88	.28 - .29 ⁽⁵⁾
Cr		.60 ⁽⁵⁾	.00 - .01 ⁽⁵⁾
Cd		.042 ⁽⁵⁾	.000 - .004 ⁽⁵⁾

(1)-(4) Refer to corresponding footnotes in Table B-4.1.

(5) Values used for further analysis in the present study.

**Table B-5.2. Estimated Annual Loading of Selected Heavy Metals
from Combined and Storm Wastewater Entering
Lake Union Directly through Outfalls**

Parameter	City Outfall Discharge ⁽¹⁾ (10 ³ Kg)	Metro Outfall Discharge ⁽²⁾ (10 ³ Kg)	Storm Drain Discharge ⁽³⁾ (10 ³ Kg)	Total Discharge (10 ³ Kg)	Final Concentrations ⁽⁶⁾ Assuming Total Mixing	
					1974 Flushing ⁽⁴⁾ (mg/l)	No Flushing ⁽⁵⁾ (mg/l)
Cu	0.21	.007	.058 - .065	.275 - .282	.000	.011 - .012
Pb	1.35	.047	1.31 - 1.38	2.71 - 2.78	.002	.111 - .113
Zn	0.78	.027	.230 - .240	1.04 - 1.05	.001	.042 - .043
Cr	0.91	.032	.000 - .008	.942 - .950	.001	.038 - .039
Cd	0.06	.002	.000 - .002	.062 - .064	.000	.003

(1)-(5) Refer to corresponding footnotes in Table B-4.2.

(6) See text relative to Table B-4.2 for explanation.

B-6. DISTRIBUTION OF SEDIMENT METALS: CORE PROFILES AND SEDIMENTATION RATES.

Since sedimentation rates in nearby Lake Washington had been found to vary greatly among sites and from one time period to another at any given site (Barnes and Schell, 1973; Schell, 1976), Metro contracted with Dr. W. R. Schell of the Laboratory of Radiation Ecology at the University of Washington College of Fisheries to evaluate the prevailing rates at the three undredged sampling stations in Lake Union (522, 526, and 532). For this purpose, the ^{210}Pb technique of Goldberg (1963) and Kiode et al. (1972) was used.

The ^{210}Pb method for dating sediments consists of measuring the ^{210}Pb radioactivity as a function of depth in core samples. The total ^{210}Pb activity measured in a core section consists of inputs from two sources: settled atmospheric fallout (unsupported ^{210}Pb) and the product of the decay of ^{226}Ra (supported ^{210}Pb). Because the supported ^{210}Pb is produced in situ, it is not representative of the product from sedimentation processes and is therefore treated as background interference and subtracted from the total. The values presented herein have all been corrected in this manner.

Sedimentation rates are estimated from the radioactive decay of unsupported ^{210}Pb . A time interval of 22.26 years (the half-life of ^{210}Pb) is assumed for sediment depth intervals separating relative ^{210}Pb activities having ratios of 2:1. This calculation is based on the assumption of a constant rate of fallout of ^{210}Pb on the water surface. The results of these measurements are presented in Fig. B-6.1. In addition to the ^{210}Pb values, stable Pb concentrations and the fractions of the dry sediments not destroyed by rigorous $\text{HClO}_4\text{-HNO}_3$ wet-ashing techniques (Smith, 1953) are also given. These fractions represent predominantly silicious material derived from erosion, as opposed to that matter originating from biological and chemical processes. The stable Pb values provide a basis for comparison of analytical quality between the laboratories of the University of Washington Radiation Ecology Section and Metro Water Quality. These data have been plotted alongside the respective Metro values in Fig. 5-1.

The sediment Pb profile from Station 526 compares fairly well with results obtained by Schell (1976) for a deep station (62 m, off Madison Park) in Lake Washington. The atmospheric fallout of unsupported ^{210}Pb is assumed to be the same for the two stations. The first increase in Pb concentration in the Lake Washington sediments occurred in 1890 when the ASARCO smelter began operation in Tacoma (Creelius and Piper, 1973), and an abrupt increase at 31 cm at Station 526 was undoubtedly due to this input. A similar increase was seen at almost the same depth in the Lake Washington core. Any significant increase of sediment Pb

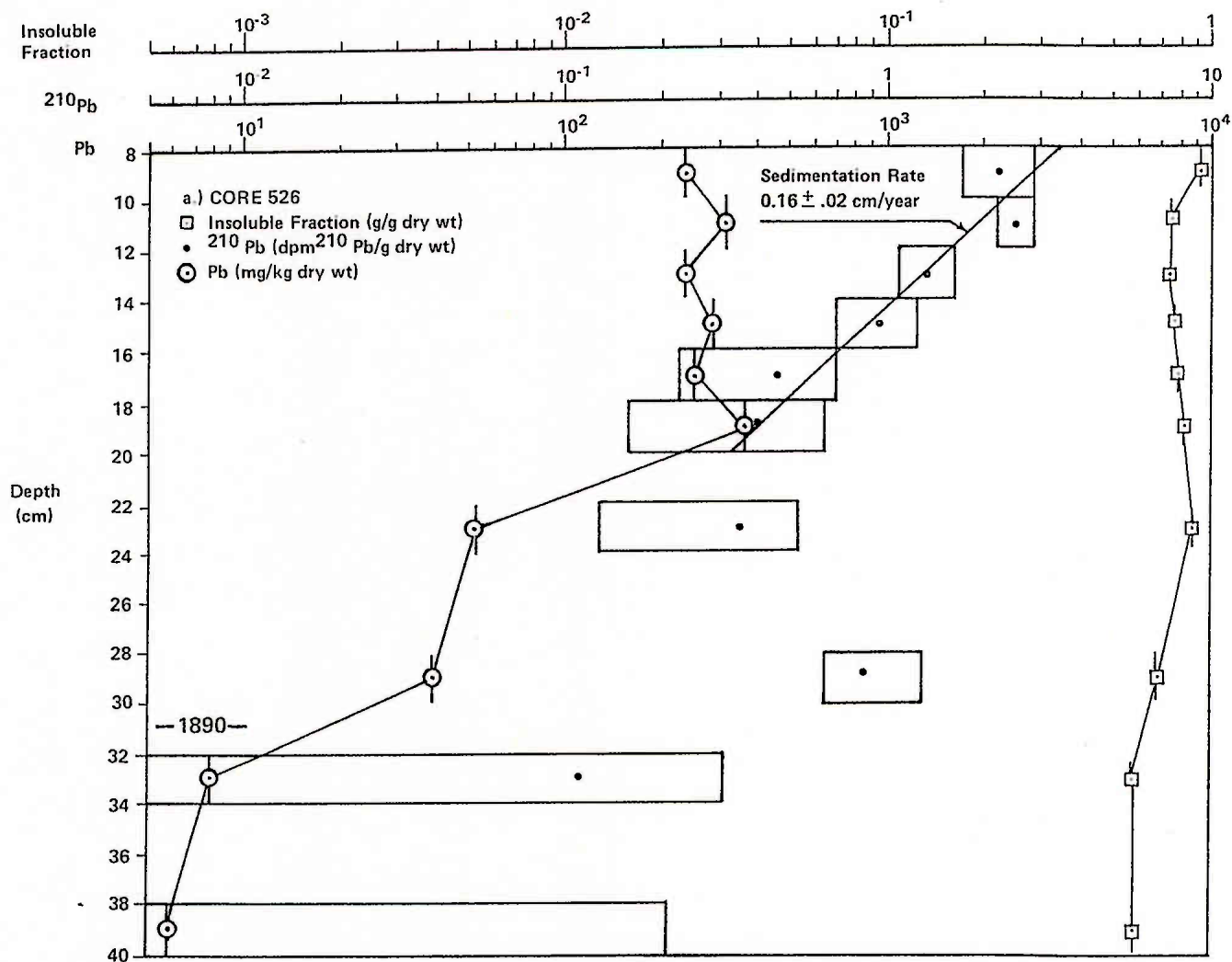


Fig. B-6.1. Profiles of lead-210, stable lead, and the insoluble fraction of the Lake Union sediments, 1974.

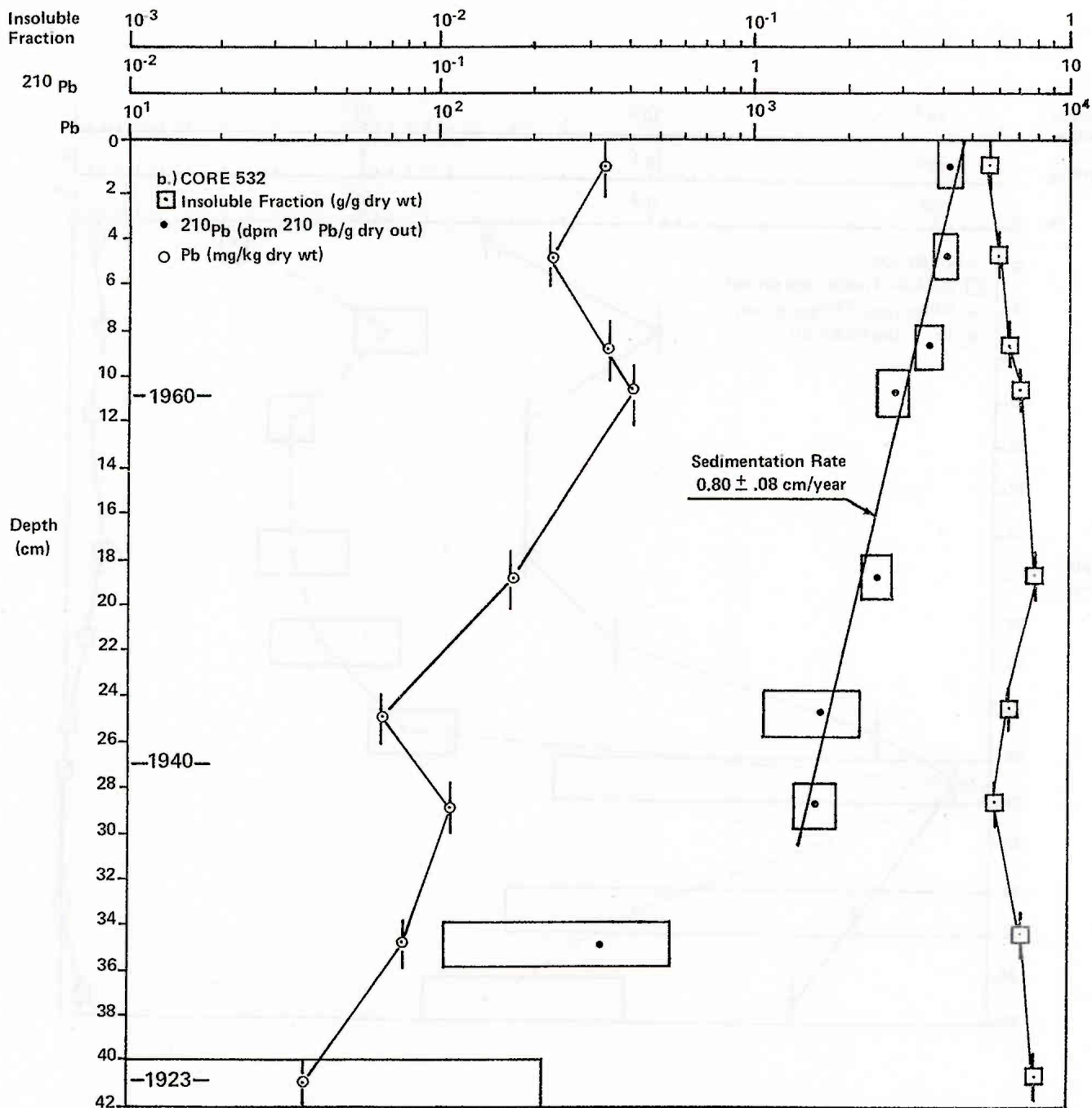


Fig. B.6.1. (continued)

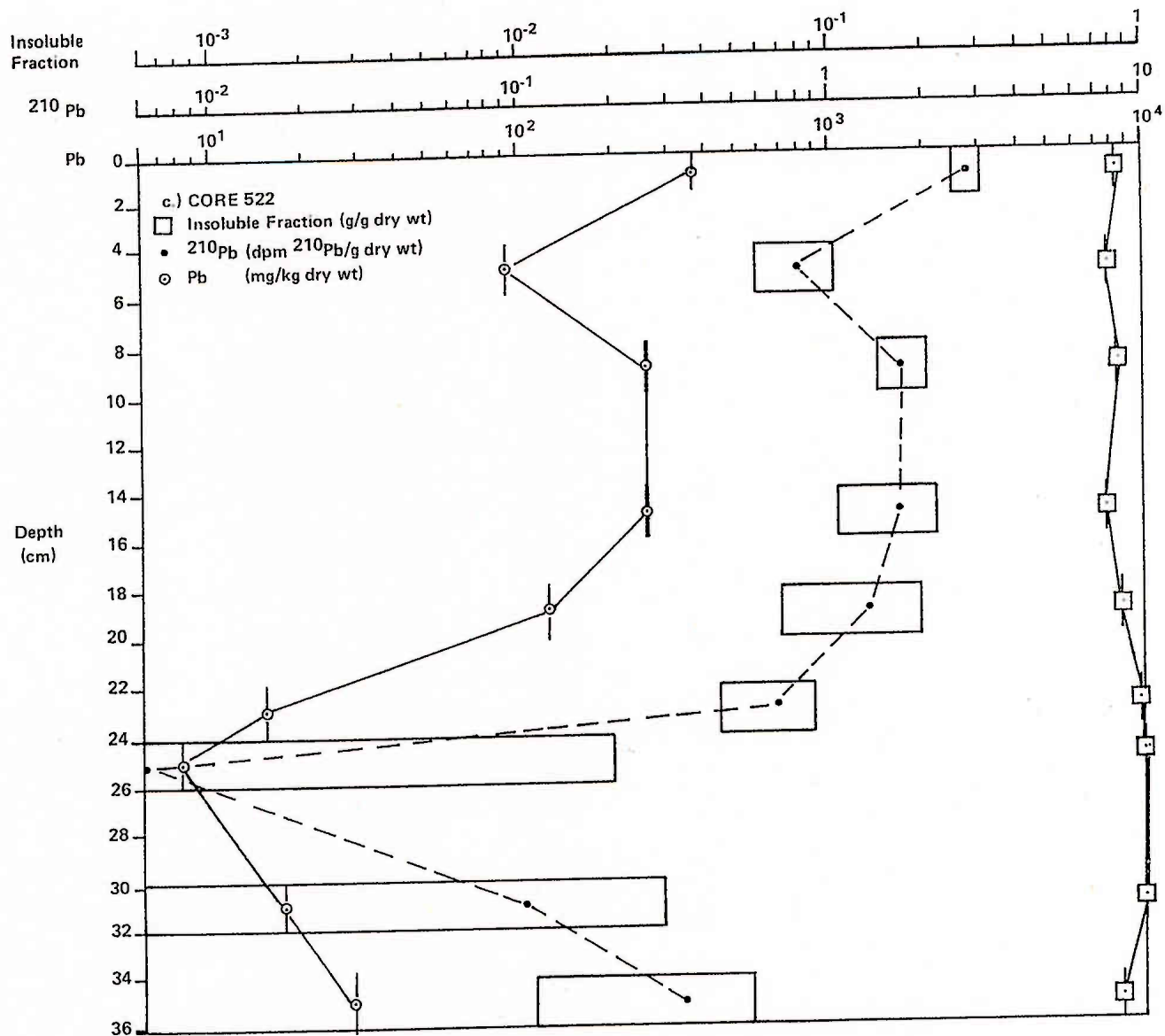


Fig. B.6.1. (continued)

concentrations above the pre-1890 concentration of ≤ 10 mg/kg dry sed. may be considered due to man's activities.

For sediment layers above those laid down in 1890, the cause-effect relationships are not as obvious. In 1913, the smelter ceased Pb production and began refining copper, but the burning of coal began contributing more Pb. Starting in the 1920's the burning of leaded gasoline by automobiles added yet more Pb.

The Pb and Cu concentrations in the sediments, however, are more than a simple sum of the metals output from human activities. These levels are effectively diluted by high inputs of erosional sediments with low-metals content. In 1916, the Hiram M. Chittenden Locks in Salmon Bay became operational, and the Cedar River was diverted to flush Lake Washington through the Ship Canal and locks system. As a result, the input of erosional sediments to Lake Union increased substantially (personal communication, W. R. Schell), and metals concentrations in sediments decreased. The sampling density for the present dating estimates of the Lake Union sediments was too light to permit a more precise definition of the depths corresponding to these historical events.

At the most, it can be said for Station 526 that no systematic change in the ^{210}Pb concentration occurred over the interval 18 to 28 cm. Above the 1890 layer, the insoluble fraction increased (Fig. B-6.1) along with the Pb concentration. This increase may be representative of a large input of material from land development. Above 18 cm, the sedimentation rate appears to have stabilized; a least-squares fitting of the ^{210}Pb data gives a value of $0.16 \pm .02$ cm/yr for the 18- to 10-cm interval, which represents approximately 50 years of the 84-year period 1890-1974.

The sedimentation pattern at Station 532 has been considerably different. The core from Station 532 which was selected for dating was the longest core taken during the study, at 42 cm, but was found not to extend to the depth of the 1890 sediments. This fact was substantiated in the interval 40-42 cm by (1) the high concentration of the insoluble fraction (0.781g/g dry sed.), (2) the high Pb concentration (37 mg/kg dry sed.), and (3) the low ^{210}Pb activity (<0.2 dpm/g dry sed.). As determined from a least-squares fitting of the data, the sedimentation rate over the interval 30-2 cm was $.80 \pm .08$ cm/yr, five times as great as that determined for the interval 18-10 cm at Station 526. On this basis, the deposit at 46 cm was laid down approximately in 1916, the year of the Cedar River diversion into Lake Washington. The stable Pb concentration showed a gradual increase from 42 cm to the surface.

The pattern indicated by the data from Station 522 is confusing. Between 36 and 22 cm the insoluble fraction was comparatively large, possibly because of silicates contributed by nearby land erosion. In the same interval, the Pb and ^{210}Pb concentrations were low, indicating dilution from inert material with low ^{210}Pb concentrations. In addition, the upper core sections contained erosional material with lower Pb and ^{210}Pb concentrations than would be expected from atmospheric input. Deposition at this site appears to be quite complicated, but it has undoubtedly been heavily influenced by land development on nearby Queen Anne Hill. Information exists on landfill being dumped into the lake near this station (A. Seymour, personal communication). On the basis of the selected core segments analyzed, it is not possible to estimate accurately dates or sedimentation rates. It can be said, however, that barring large-scale sediment mixing processes, the sediments above 36 cm were all laid down after 1890. This conclusion is substantiated by relatively high stable Pb values between 36 and 30 cm.

Station 518 was dredged in 1954 and was therefore not included in the analysis. However, this fact enables a simple calculation of sedimentation rate to be made with some confidence. Between 1954 and 1974, 20 cm of sediment was deposited, at an average rate of 1 cm/yr. This value compares favorably with that ($.80 \pm .08$ cm/yr) determined by alpha spectrometry for the top 30 cm of the sediments at the other channel station, 532. Moreover, the concentrations of all six metals analyzed (Fig. 5-1) show that the dredging removed (at least) all sediments deposited since 1890.

B-7. SEASONAL TRENDS IN ABUNDANCE OF COLIFORMS IN THE LAKE WASHINGTON SHIP CANAL.

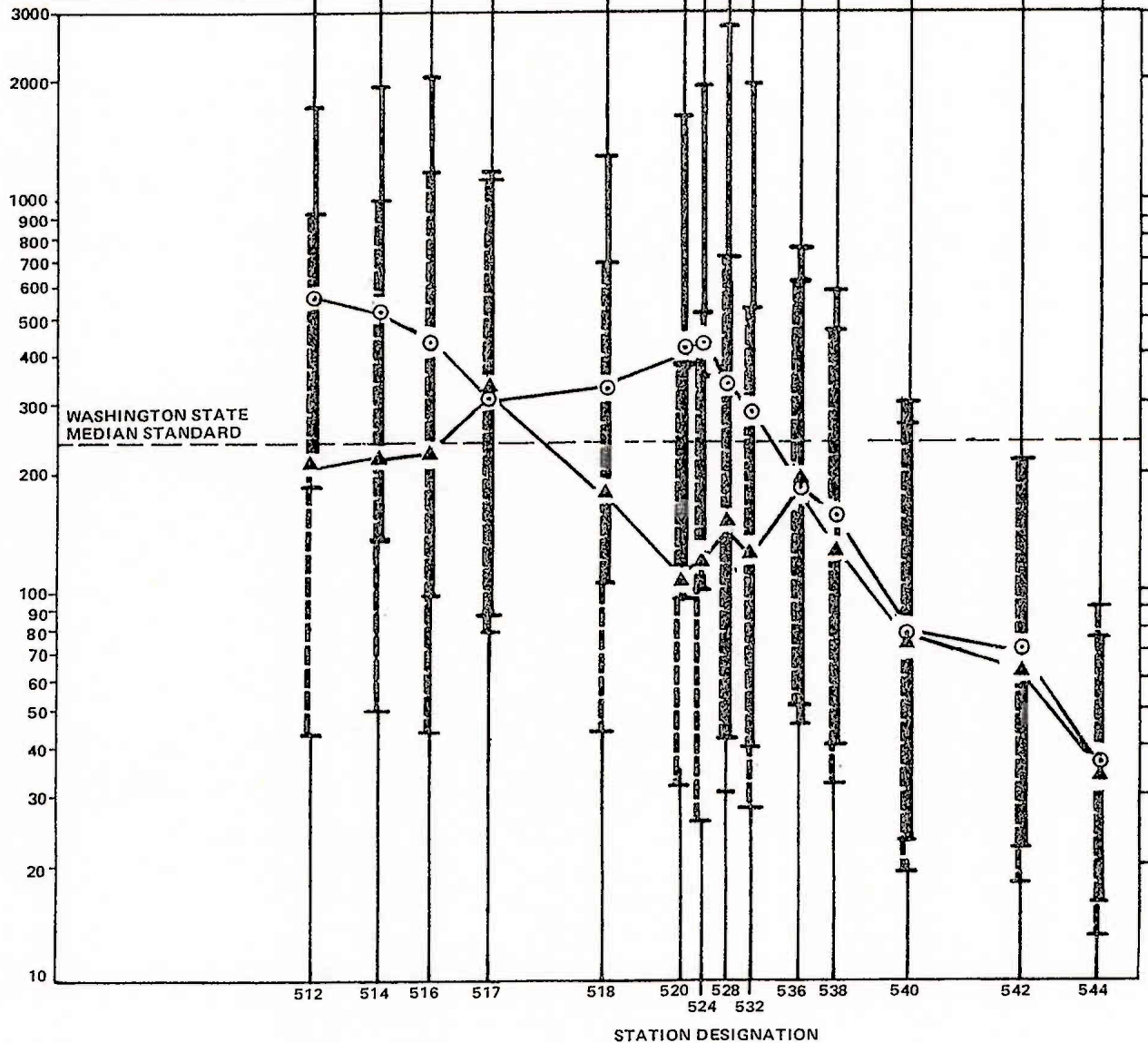
On the basis of 12-year trends (Fig. 4-16), the data for the period 11/69 - 12/75 were selected as statistically representative of the abundance of total coliforms in Lake Union and the ship canal. The data for the winter/wet season were separated from those for the summer/dry season. The geometric means and attendant standard deviations were calculated for these two sets of data for the 14 sampling stations occupied monthly (in some cases semimonthly) during this period. The results are presented as a function of geographic distribution in Fig. B-7.1.

In general, the total values for coliforms increased as a function of distance between Lake Washington and Puget Sound, regardless of season. The increases were undoubtedly related to the cumulative influence of sewage and stormwater overflows (Fig. B-4.2) with the seaward movement of water through the system; it would appear that the flow reversals seen by Driggers (1964) and Layton (1975) were not prevalent. This analysis is further substantiated by the significantly lower lake-to-sound increase in total coliforms during the summer/dry season. Overflows during this period were considerably less frequent than during the months of higher rainfall. Between Lake Washington and the University Bridge (Station 536), the summer and winter means were quite similar, with one doubling of total coliform counts between Lake Washington and the Montlake Bridge (Stations 544 to 540), and another between the Montlake Bridge and the University Bridge.

Two stations in the ship canal are atypical of the general total coliform trends. The counts for the dry season at Stations 517 and 536 are inconsistently high, whereas the counts for the wet season at Station 517 are surprisingly low. At both locations the means for the wet and dry seasons are virtually identical, so that inputs other than storm or combined sewer overflows are implied. Station 517 is the only standard sampling site at which the mean count of total coliforms exceeded the Washington State median standard the year round.

Fig. B-7.1. Geometric means of historical counts of total coliforms from Lake Union and the Lake Washington Ship Canal.

GEOMETRIC MEAN OF THE TOTAL COLIFORM COUNT/100mls



B-8. A HYPOTHESIS FOR THE DISTRIBUTION OF NONPARTICULATE CONTAMINANTS FROM WASTEWATER AND SURFACE RUNOFF

To further clarify the relationship between levels of total coliforms and station location, we replotted the data in Fig. B-7.1 on a linear scale, as a function of distance along the main channel (Fig. B-8.1). If we except the data from Stations 520, 524 and 528 as outside of the principal flow, and the data from Stations 517 and 536 as subject to uncharacteristic influences, we may satisfactorily represent the remaining data for the two seasons by straight lines, as shown. The relationships present a strikingly consistent key to the mechanics of water contamination in the ship canal system.

The linearity implies that any quasiconservative dissolved or suspended contaminant characteristically introduced in appreciable quantities by wastewater overflows and/or direct surface runoff should show a linear increase in water moving from Lake Washington to Puget Sound. That the pertinent contaminant should be at least quasiconservative (not appreciably affected by biological processes on a short-term basis) must be stressed here. For example, although the levels of nutrients needed for algal production would be expected to increase linearly under high flow conditions (generally commensurate with low temperatures, low light, and short residence times), they would be more subject to algal uptake (and therefore greater variability) under low flow conditions. In either case, it is critical to note that potential availability of nutrients would increase linearly on a long-term statistical basis, as noted. Further discussion on nutrient concentrations in light of this relationship may be found in Section B-10 of this report.

A discussion of the significance of this pollutant-additions model to the levels of the various contaminants observed in Lake Union seems pertinent at this point. Although the x-axis locations of Stations 520, 524, and 528 in Fig. B-8.1 are indefinite owing to insufficient knowledge of the lake circulation, the relevant data on coliforms have been entered between those values representing the lake inlet and outlet stations. It is apparent that the data from Station 528 are not grossly out of keeping with the two linear relationships. Thus, it would seem that this station is more strongly influenced by the through-lake circulation than the other two are. Levels of total coliforms at Stations 520 and 524, however, are appreciably higher during the wet season and lower during the dry season than expected.

This phenomenon appears to be due to a longer residence time for water in this portion of the lake. In the mainstream flow through the ship canal, the mixed concentration of a contaminant depends on its initial concentration and the

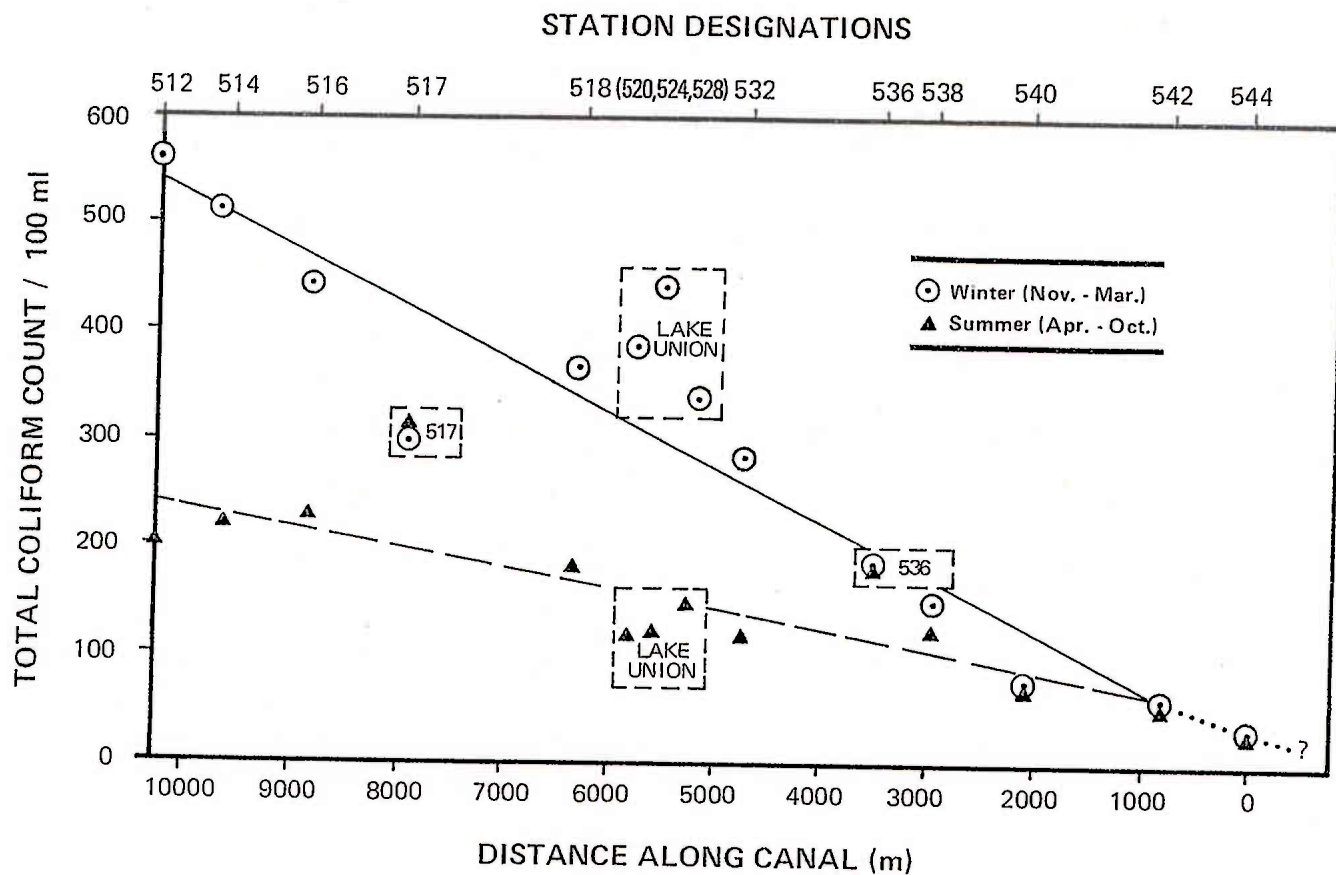


Fig. B-8.1. Mean counts of total coliforms in winter and summer vs distance along the Lake Washington Ship Canal and through Lake Union, for the period 1969-1975 (Metro data).

ratio of its input rate to the rate of water flow through the system (i.e., total residence time for a given water mass). For water entering the southern portion of Lake Union, the flow rate is decreased and a disproportionately large volume of contaminants is added by overflows and runoff in that area. The probable result is a large increase in pollutant concentration during wet weather and a small or negligible increase during dry weather. The decrease in coliforms during dry weather observed in the present study probably reflects the sterilizing influence of solar radiation (Whitmore et al., 1969). The implication of flow stagnation and pollutant buildup in Lake Union is graphically supported by data on sediment quality, presented in Section 5.

It is important to stress that the foregoing discussion was based on counts of coliforms at the surface only. Observations by Layton (1975) indicate that outfall outlets positioned below a density interface will discharge pollutants that are subsequently trapped below the gradient structure. Thus, the present observations would be expected to apply to all depths only during periods of minimal stratification (November-April). Relationships near the bottom for the rest of the year remain unknown. It is expected, however, that the saltwater intrusion, which moves countercurrent to that flow postulated above as predominant, may complicate the picture considerably.

B-9. A HYPOTHESIS FOR THE DISTRIBUTION OF PARTICULATE CONTAMINANTS FROM WASTEWATER AND SURFACE RUNOFF

As indicated by Table 5-2, the only sites in the ship canal system having appreciable concentrations of oil and grease of petroleum origin are the Lake Union Stations: 522, 526, and 532. Since the Metro Galer Street storm outfall is only 150 m due south of Station 522, it is of interest to calculate the ratio of petroleum-based to animal- or vegetable-based oils and greases in typical storm overflows. A representative mean concentration for oils and greases in commercial zone runoff is 11 mg/l (Farris, 1974), of which the major fraction is of petroleum origin. A representative mean figure for oils and greases in domestic sewage is 26 mg/l, of which 73 percent is of animal and vegetable origin (Hunter and Heukelekian, 1965). If we consider 6:1 to be a representative ratio of mean storm flow (combined wastes) to mean base-line flow (sanitary wastes) (Edward Cox, personal communication), then it follows that the ratio of petroleum-based to animal- or vegetable-based oils and greases discharged from the Galer Street storm outfall is 3.3:1 for combined wastewater with a mean total oil and grease concentration of 13.5 mg/l. That this ratio differs significantly from that measured for the sediments at Station 522 (1:1) is probably largely due to the natural separation of the particulate from the colloidal and dissolved fractions through settling processes. Applying Hunter and Heukelekian's value of 43 percent for the mean settleable fraction in domestic sewage, we derive an estimate of 9 percent for the mean settleable fraction of oils and greases in the discharged stormwater. This relatively low figure is further supported by the observation that stormwater runoff in commercial areas characteristically has a higher visible concentration of surface oils than domestic sewage does.

Since combined and storm wastes are the primary source in Lake Union and the ship canal of oils and greases of petroleum origin, it is of interest to note that the sediment levels of this contaminant are relatively low outside of Lake Union. On the other hand, the sediment levels of animal- or vegetable-based oils and greases in Lake Union, Salmon Bay (M2), and Portage Bay (M4) are high.

These observations may be important clues to the mechanism of particulate contamination in this system. Whereas the effect of decreased flow velocities on dissolved and colloidal pollutants is increased concentration (from overflow additions), there is a major secondary influence on suspended particulate contaminants. They may settle into the sediments and be removed from further circulation. Thus, in the wide sections of the canal in which flow rates decrease, the highest sedimentation rates can be expected. Whether or not a concurrent buildup of pollutant concentrations is observed depends on the relative proportions of contaminated and noncontaminated particles in the bedload.

Accordingly, the increases of animal- or vegetable-based oils and greases observed in Lake Union, Salmon Bay, and Portage Bay indicate either a predominance of contaminated particles or a higher settleability, or both. An increased production of such particulate matter during summer-low flow conditions would satisfy both criteria. Such is provided by the seasonality of the phytoplankton blooms and (assumed) zooplankton peaks. Thus, it seems likely that the high sediment contents of oils and greases (and COD/TVS) measured in these three areas are contributed in large part by the biota. This conclusion does not seem unreasonable in light of the findings of Ketchum and Redfield (1949) of a lipid content of 20-25 percent of the total organic dry weight of six different species of freshwater algae. This amount is high compared with that found in land plants.

On the other hand, the petroleum-based oils and greases are apparently added to the system predominantly during high-flow conditions, by wastewater overflows. Under these conditions, Lake Union may be the only embayment with still enough bottom waters to permit appreciable rates of sedimentation.

As previously mentioned, animal- or vegetable-based oils and greases are also introduced by the overflows, a source that was assumed predominant at Station 522 for the purpose of the above calculations. Even with the close proximity of the Galer Street outfall, however, the net annual fraction added by the lake biota may be sufficiently large to demand consideration in such computations. If so, the estimate of 9 percent derived for the mean settleable fraction of oils and greases in the stormwater is too low (for example, if 25% of the observed content of animal- or vegetable-based oils and greases of the sediments was contributed by the biota, the mean settleable fraction in the stormwater would be 14%). More extensive spatial and temporal sampling of the sediments would serve to reduce the degree of speculation in these ruminations.

B-10. TROPHIC CLASSIFICATION OF LAKE UNION

Eutrophication, the addition of nutrients and its consequences (EPA, 1973), is the process by which a lake reaches a higher trophic state. The process represents the cumulative effects of natural aging and/or man's activities; the latter cause is commonly referred to as cultural eutrophication. Eutrophic lakes characteristically have very high nutrient levels and often produce excessive aquatic plant growth; thus their recreational and aesthetic values can be adversely affected.

Since Lake Union is located in the heart of a metropolitan area, it is a prime candidate for cultural eutrophication. For the present study, we have attempted to answer the following questions: How eutrophied is Lake Union in terms of U. S. and world classifications? Does the apparent trophic state vary spacially within Lake Union? How does the condition of the lake compare with that of others in the same region?

Because lakes vary so much in climate, hydrology, morphometry, basin soil type and water, and land use, no single analytical criterion has been established for estimation of their trophic state. The usual method for detecting eutrophication and measuring its effects involves the synthesis of many different parameters (indices) into a general pattern, which denotes the average trophic status of the body of water in question (Lee, 1970; Bortleson et al., 1974).

Representing a synthesis of many different factors, the concentration, duration, and generic composition of a phytoplankton population are often used to provide trophic state estimates (Rawson, 1956; Vollenweider, 1968; EPA, 1973). Vollenweider feels that one of the most universal indices of the nutrient state of a body of water is the maximum density of its phytoplankton populations. Hence, he has set up the following trophic classification scheme:

<u>Class</u>	<u>Phytoplankton Cell Density (cm³/m³ water)</u>
Ultraoligotrophic	<1
Mesotrophic	1-5
Eutrophic	>3
Highly Eutrophic	>10

By this measure, all of the Lake Union sampling stations except 532 would be considered highly eutrophic (Table B-10.1) and Station 532 on the borderline between eutrophic and highly eutrophic. It seems probable that the maximum densities among the four stations do not differ significantly.

Table B-10.1. Summary of Trophic Indices of Major Lakes in the Lake Washington Drainage Basin

Lake	Reference	Study Date	Station	Maximum Spring, Surf [Ortho PO ₄] (mg/l)	Maximum Spring, Surf. [Inorg. N] (mg/l)	Maximum Phytopl. Density (cm ³ /m)	Rel. Freq. Phytopl. Blooms (%)	Dominant Summer Phytopl. Species	Mean Summer [Chl. a] (mg/l)	Rel. Bottom O ₂ (<4 mg/l)	Rel. Freq. (%)
Union	Collins & Seckel (1954)	1951	8.5 (522)	.029							100
			10 (near 532)	.040							64
		1952	8.5								100
Union	Present Study	1974	518	.060	.362	10.8	15	<i>Lyngbya limnetica</i>	3.4	0	0
			522	.060	.371	12.6	35	<i>Anabaena</i> sp.	3.8	38	38
			526	.060	.351	15.2	27	<i>Lyngbya limnetica</i>			
Sammamish	Emery (1972)	1970	532	.040	.373	9.4	25	<i>Anabaena verrucosa</i>	3.4	12	12
			general	.012	.68	40	35	<i>Lyngbya limnetica</i>	3.8	31	31
								<i>Aphanocapsa</i> sp.	2.5	50	50
Washington	Pieterse (1974)	1971	general	.012	.52	20	35	<i>Coelosphaerium</i> sp.			
								<i>Microcystis</i> sp.			
		1973	Madison Park					<i>Aphanizomenon</i> sp. (same as above)	2.5	38	38
Washington		1974	Madison Park	.020	.375	2.9	0	<i>Anabaena verrucosa</i>	2.1*	27	27
								<i>Lyngbya limnetica</i>	3.5*		

* W. T. Edmondson, unpublished data.

The data on nutrients appear to substantiate Vollenweider's classification. Sawyer (1947) has stated that a body of water has the potential to produce bloom and nuisance levels of phytoplankton if the spring maxima of assimilable phosphorus (orthophosphate) and inorganic nitrogen (nitrate, nitrite, and ammonia) exceed .010 mg/l and .300 mg/l, respectively. Vollenweider (1968) goes one step further by saying that these nutrient levels indicate when a lake is "in danger with regard to its trophic level." Nutrient concentrations observed at all of the Lake Union stations exceed these values (Table B-10.1).

Compared with 1973-1974 data for other small mesotrophic-eutrophic lakes in the Lake Washington drainage basin (Uchida et al., 1976) and with 1970-1971 data for Lake Sammamish (Emery, 1972), the ortho PO_4P concentrations are slightly high, whereas the inorganic nitrogen concentrations are low (Table B-10.1). On the other hand, compared with the Lake Washington 1974 spring nutrient maxima (Pieterse, 1974), the orthophosphate concentrations are high, but the inorganic nitrogen concentrations are very similar. It would appear that Lake Union and Lake Washington have a similar or common pool of inorganic nitrogen, whereas a significant fraction of the orthophosphate in Lake Union comes from sources within the lake itself. Wastewater overflows and the seasonally anoxic hypolimnion are the most probable origins of this additional phosphorus.

Orthophosphate concentrations in Lake Union in 1974 were somewhat higher than in 1951. At that time Collias and Seckel (1954) measured spring maximum ortho $\text{PO}_4\text{-P}$ concentrations of .029 and .040 mg/l for the surface waters at Stations 522 and 532, respectively. Thus, a 50 percent increase occurred at Station 522 and no change at Station 532. With regard to the levels of assimilable phosphorus, Lake Union seems to have deteriorated only slightly in the last 25 years. On the basis of these data and Sawyer's (1947) criteria, conditions appear to have been conducive to phytoplankton blooms in 1951 also.

One of the more direct means of estimating the trophic state of more severely eutrophied waters is to note how much of the year blooms occur, producing excessive amounts of phytoplankton (Lee, 1972). A bloom is commonly considered to consist of at least $0.5\text{-}1.0 \times 10^6$ cells/l, a somewhat arbitrarily selected density that corresponds fairly well to Vollenweider's (1968) eutrophic threshold criterion of $3 \text{ cm}^3/\text{m}^3$. On this basis, blooms were present in Lake Union at all stations for 15 to 35 percent of the year 1974. Blooms were least frequent at Station 518 (15%) and occurred most often at the in-lake stations (526: 27%, 522: 35%). Of the other small mesoeutrophic lakes in the Lake Washington and Green River drainage basins, Lakes Crystal, Meridian, Sawyer, Serene, Star, and Stickney

are of a comparable degree of eutrophication and Lake Ballinger is of a somewhat greater one (44% bloom occurrence) (Uchida et al., 1976). The more eutrophied lakes in this region (Cottage, Desire, Pine, and Wilderness) had blooms about twice as often as Lake Union during 1974. Lake Sammamish in 1970 - 1971 (Emery, 1972) had blooms with an equal frequency. Lake Washington during 1973 (Pieterse, 1974) had no bloom concentrations of phytoplankton.

As previously mentioned, various species of blue-green algae were responsible for observed increases in cell density in Lake Union during the summer months (June 30-September 30). A comparison of Figures 6-1 and 6-2 indicates that these pulses were not as apparent in terms of chlorophyll a concentrations. Uchida et al. (1976) also noted this discrepancy for Lake Wilderness in 1974. It would appear, therefore, that chlorophyll a concentration is not a good measure of blue-green algal biomass.

District X of EPA has recently begun to use chlorophyll b and c, in addition to a, as indices to biomass estimates (R. Kreizenbeck, personal communication). The b and c pigments have been found to be particularly important for evaluation of blue-green algal blooms. Having no chlorophyll b and c data for Lake Union, we can only present the available chlorophyll a values (Table B-10.1), which have been used by EPA workers and other investigators as a measure of trophic level during the summer months. As none of the stations had a chlorophyll a concentration exceeding 4 mg/l, Lake Union would be considered oligotrophic by EPA (1973) standards (< 4 mg/l: oligotrophic, 4-10 mg/l: mesotrophic, >10 mg/l: eutrophic). This estimate is obviously not in keeping with the previously discussed estimates.

On the basis of the predominant limnetic phytoplankton for the summer period (Table B-10.1), the approximate trophic level of a lake can be determined. Rawson (1956) selected phytoplankton species having the following characteristics as indicators of the highest level of eutrophy: "large filamentous or colonial blue-green algae having gas vacuoles for floatation." Although blue-green algae are dominant in Lake Union during summer, they are primarily of the very finely filamentous species Lyngbya limnetica. Its population densities barely reach bloom levels during the summer. Only the in-lake station, 522, fits Rawson's criterion; the summer populations of Anabaena sp. (thought perhaps to be A. circinalis) are short-lived but quite dense at that location. At Station 526, heavy blooms of Anabaena verrucosa are also produced during summer, but the filaments of this species are shorter and nonvacuolated.

Measured by Rawson's criterion, the 1974 Lake Union sampling stations would have the following trophic classifications: 522 - mesoeutrophic; 518, 526, and 532 - mesotrophic.

Station 522 was not classified as fully eutrophic because the Anacystis sp. at Station 522 had very small cells, only 1 μ m in diameter. It was found not to be Anacystis cyanea Kuetzing, emend. Drouet and Daily (1956) { (Microcystis aeruginosa Kuetzing, emend. Elenkin (1924)) }, which is considered to be a primary indicator of eutrophication (Rawson, 1956).

In general, on the basis of the various phytoplankton indices, one would classify the lake as mesoeutrophic. The deep, in-lake station, 522, seems to have the highest phytoplankton densities, and the canal stations, 518 and 532, seem to have the lowest. The flushing action of water movement through the ship canal may help to disperse potential blooms.

B-11. PREY-PREDATOR RELATIONSHIPS FOR SOME OF THE REGIONAL AQUATIC FAUNA

Having previously discussed the zooplankton, benthic macrofauna and fishes separately, we feel it worthwhile also to document some of the information available on their food chain relationships. These data are summarized in Table B-11.1. For the fish species listed, the ten most important food organisms according to numbers ingested are (in descending order and weighted by percentage of total diet): dipteran larvae and pupae, mysid shrimp, gastropods, chironomids, sculpins, cladocerans, copepods, juvenile sockeye salmon, crayfish, and trichopteran larvae. On the basis of taxonomic or ecological groupings, the prey organisms may be ranked in importance as follows: (1) insect larvae and pupae (chironomids, dipterans, hemipterans, trichopterans), (2) planktonic crustaceans (copepods, cladocerans, mysids), (3) small fish (sculpins, juvenile sockeye salmon, sticklebacks) (4) mollusks (gastropods, pelecypods), and (5) benthic crustaceans (amphipods, crayfish, isopods).

Table B-11.1. Diet Composition of Selected Fish Species in the Lake Washington Drainage Basin

<u>Fish Species</u>	<u>Study</u>	<u>Sampling Locale</u>	<u>Food Organisms</u>	<u>Percent of Total Diet (by Number)</u>
Yellow Perch	Costa, 1973	Lake Washington	Mysid shrimp (<i>Acanthomysis awatchensis</i>)	69.1
			Chironomids (<i>Chironomus</i> spp.)	17.0
			Sculpins (<i>Cottus asper</i>)	8.0
			Insect larvae (Trichopterans)	4.2
			Mollusks	0.5
	Holland, 1952	Portage Bay, Ship Canal	Leaches	0.2
			Sticklebacks (<i>Gasterosteus aculeatus</i>)	0.1
			Mysid shrimp	23.3*
			Chironomids	7.6*
			Dipteran larvae	2.9*
Black Crappie	Brown, 1968	Union Bay	Crustacean larvae	2.7*
			Dipteran larvae and pupae	78.8
			Copepods	12.6
			Cladocerans	4.2
			Amphipods	2.5
		Ship Canal	Algae	1.2
			Dipteran larvae and pupae	48.0
			Cladocerans	35.2
			Copepods	15.2
			Amphipods	0.5
Brown Bullhead	Imamura, 1975	Union Bay	Isopods	ND
			Crustaceans	ND
			Chironomid larvae	ND
Northern Squawfish (adults)	Olney, 1975	Lake Washington	Sculpins	34
			Juvenile sockeye	18
			Crayfish	15
			Snails	ND
Northern Squawfish (juveniles)			Small crustaceans	predom.
			Insects	predom.
			Cladocerans	ND
			Copepods	ND
			Chironomid larvae and pupae	ND
Peamouth	Nishimoto, 1973	Lake Washington	Gastropods	47.1
			Chironomid larvae and pupae	28.6
			Isopods	10.7
			Insect larvae (Trichopterans)	7.6
			Amphipods	3.2
			Pelecypods	1.6
			Cladocerans	1.0
			Hydracaridans	1.0
			Insects (Hemipterans)	<0.1
			Fish eggs	<0.1

* Percent by occurrence
 ND: No data
 predom: Predominant food item